PugetSoundScienceUpdate

Draft April 12, 2011

PugetSoundPartnership
our sound, our community, our chance

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Editor’s note

The Puget Sound Science Update is a represents the state-of-the-science supporting the work of the Puget Sound Partnership to restore and protect the Puget Sound ecosystem. The Puget Sound Science Update represents an advancement in the development and use of science to support Puget Sound recovery in two important ways. First, the content of the Puget Sound Science Update was developed following a process modeled after the rigorous peer-review process used by the Intergovernmental Panel on Climate Change (IPCC), in which small author groups produced draft assessment reports synthesizing existing, peer-reviewed scientific information on specific topics identified by policy leaders. These drafts were peer-reviewed before the final reports were posted. Second, the Puget Sound Science Update will be published on-line following a collaborative model, in which further refinements and expansion occur via a moderated dialog using peer-reviewed information. Content eligible for inclusion must be peer-reviewed according to guidelines.

In the future, there will be two versions of the Update available at any time:

(1) a time-stamped document representing the latest peer-reviewed content (new time-stamped versions are likely to be posted every 4-6 months, depending on the rate at which new information is added); and

(2) a live, web-based version that is actively being revised and updated by users.

The initial Update you see here is a starting point to what we envision as an on-going process to synthesize scientific information about the lands, waters, and human social systems within the Puget Sound basin. As the document matures, it will become a comprehensive reporting and analysis of science related to the ecosystem-scale protection and restoration of Puget Sound. The Puget Sound Partnership has committed to using it as their ‘one stop shopping’ for scientific information—thus, it will be a key to ensuring that credible science is used transparently to guide strategic policy decisions.

The Update is comprised of four chapters, and you will note that some are still at earlier stages of completion than others. Over time—through the process of commissioned writing and user input through the web-based system—the content of all four chapters will be more deeply developed. We are relying in part on the scientific community to help ensure that the quality and nature of the scientific information contained in the Update meets the highest scientific standards.

Preface

Who are the authors of the Puget Sound Science Update?

Leading scientists formed teams to author individual chapters of the Puget Sound Science Update. These teams were selected by the Puget Sound Partnerhship's Science Panel in response to a request for proposals in mid-2009. Chapter authors are identified on the first page of each chapter. Please credit the chapter authors in citing the Puget Sound Science Update.

What are the Puget Sound Partnership and the Science Panel?

Please visit psp.wa.gov to learn about The Puget Sound Partnership.

Please visit science panel web page to learn about the Science Panel.

Has the Puget Sound Science Update been peer reviewed?

The original chapters of the Puget Sound Science Update were subjected to an anonymous peer review refereed by members of the Puget Sound Partnership's Science Panel. Reviewers are known only to referees on the Science Panel and the Partnership's science advisor.

What is "content pending review"?

The future web presentation is intended to offer a venue for updating, improving, and refining the material presented in the Puget Sound Science Update. Suggested amendments and additions are presented as "content pending review" on each page when an editor, perhaps working with a collaborating author, has developed some new content that has not yet been formally adopted for incorporation into the section. As "content pending review," this content should not be cited or should be cited in a way that makes clear that it is still in preparation.

How can I contribute new material to the Puget Sound Science Update?

Please visit the Puget Sound Partnership website to learn about how you can help improve, update, and refine the Puget Sound Science Update, or send an e-mail to pssu@psp.wa.gov to get the process started.

How can I cite the Puget Sound Science Update?

We recommend citations this version in the following format:

[Authors of specific chapter or section]. April 2011. [Section or chapter title] **in** Puget Sound Science Update, *April 2011* version. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

"Content pending review" of the Puget Sound Science Update has not been fully reviewed for publication. If you elect to cite this information, we recommend that you contact the named author(s) to cite as a personal communication or cite the web-presentation using the following format:

[Authors of pending material]. In prep. Content pending review presented in *[Section or chapter title]* **in** Puget Sound Science Update. Accessed from http://www.psp.wa.gov/. Puget Sound Partnership. Tacoma, Washington.

Chapter 1A. Understanding Future and Desired System States

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Introduction

The Puget Sound Partnership (PSP) is charged with the task of reversing the decline in the ecological condition of Puget Sound and restoring its health by 2020 [1]. Since the creation of the PSP and the publication of the Puget Sound Partnership Action Agenda, the Puget Sound ecosystem has become a national example of implementation of ecosystem-based management (EBM; [2]). As the Puget Sound region considers the dozens of near-term actions for ecosystem recovery, policy makers, resource managers, and scientists must be able to answer two key questions about the state of the ecosystem: 1) where are we going?, and 2) how do we know when we get there? Answering the question of what constitutes a healthy Puget Sound requires a thoughtful articulation of what the future of Puget Sound should be and scientifically rigorous means for measuring progress towards this desired future. This is the aim of this chapter.

Terminology and Concepts **Open Standards**	Open Standards for the Practice of Conservation , a set of adaptive management steps developed by the Conservation Measures Partnership as a framework for planning and implementing conservation action. The Open Standards methodology is being used by the PSP to put the Action Agenda into a performance management framework.
Results Chain	One component in the Open Standards framework being used by the PSP. A tool showing how a particular action taken will lead to some desired result. Diagrams link short-, medium- and long-term results in "if... then" statements. The three basic elements are a strategy, expected outcomes, and desired impact.
Management Strategy Evaluation	(MSE) Conceptual framework that enables the testing and comparison of different management strategies designed to achieve specified management goals
Performance Management	A system to track implementation and communicate progress of a conservation project or program

For more information and links to references, see Glossary

A properly designed monitoring program is essential for determining progress towards a desired future ecosystem state. Monitoring encompasses the routine measurement of ecosystem indicators to assess the status and trends of ecosystem structure and function. Successful monitoring requires consideration what we should monitor and why we are monitoring it. Broadly, there are two goals for monitoring in the Puget Sound ecosystem. The first goal is to monitor status and trends of the ecosystem. This may take the form of snapshots of specific regions, or, more usefully, status monitoring tracks variability in carefully selected indicators over time. Status monitoring is fundamentally concerned with documenting spatial and temporal variability in ecosystem components and thus ideally relies on consistent long-term monitoring in a network of sites.

A second aim of monitoring is to evaluate the effectiveness of management strategies. Effectiveness monitoring thus aims to detect changes in ecosystem status that are caused by specific management actions. Effectiveness monitoring is ideally informed by a conceptual or numerical system model. Such models can be used to generate predictions or hypotheses of how management actions might shift the system towards a desired state. A carefully crafted plan for effectiveness monitoring requires indictors of 1) compliance with regulations; 2) ecosystem pressures (the object of management action); and, 3) status of the ecosystem affected by these pressures. Such a plan for effectiveness monitoring allows a determination of how well predictions about appropriate management strategies performed, and provides a formal means for learning about the system and how management actions influence the system.

In the 2008 Action Agenda, the PSP established five priority strategies, one of which includes developing a performance management system to track and assess progress towards an ecologically healthy Puget Sound [1]. To this end, the PSP has adopted the Open Standards for the Practice of Conservation ("Open Standards"[3]) as a framework for implementing and tracking the progress of the Action Agenda. The Open Standards describe steps in the design, implementation and monitoring of conservation projects, two components of which are the identification of ecosystem components and indicators for those components; and development of "Results Chains," diagrams that map specific management strategies to their expected outcome (e.g., reduction of a threat) and their impact on key components of the ecosystem using a series of "if...then" statements [4]. The Open Standards is thus a tool that can be used to articulate "where we want to go", and inform both status and effectiveness monitoring to determine if we reached our goal.

In this section of the Puget Sound Science Update (PSSU), we first critically review published reports that describe desired future states of the Puget Sound ecosystem, and suggest ways to incorporate new information generated by such future visions into the results chain model. We next introduce a flexible framework for selecting indicators of the biophysical components of the ecosystem (the human components are addressed in Section 1B of this document, 'Incorporating Human Well-Being into Ecosystem-Based Management'), and establish transparent criteria for judging an indicator's ability to reliably track changes in ecosystem status. Using these criteria, we then provide an evaluation of 270 candidate ecosystem indicators. Finally, we review targets and benchmarks for ecosystem indicators in Puget Sound; where they are found wanting, we describe a number of approaches that could be applied to scientifically inform the development of management targets and benchmarks. It should be noted here that while the PSP and the authors of this document consider the Puget Sound ecosystem to be inclusive of humans, this section develops indicators for the biophysical components of the ecosystem, and therefore in those sections, the term "ecosystem" refers exclusively to the biophysical components.

Ecosystem Health

Rapport and colleagues (1985) suggested that the responses of stressed ecosystems were analogous to the behavior of individual organisms [5]. Just as the task of a physician is to assess and maintain the health of an individual, resource managers are charged with assessing and, when necessary, restoring ecosystem health. This analogy is rooted in the organismic theory of ecology advocated by Clements over 100 years ago, and is centered on the notion that ecosystems are homoeostatic and stable, with unique equilibria [6]. In reality however, disturbances, catastrophes, and large-scale abiotic forcing create situations where ecosystems are seldom near equilibrium. Indeed, ecosystems are not "superorganisms"—they are open and dynamic with loosely defined assemblages of species [7]. Consequently, simplistic analogies to human health break down in the face of the complexities of the non-equilibrial dynamics of many ecological systems [8]. Even so, the phrase "ecosystem health" has become part of the lexicon of EBM and resonates with stakeholders and the general public [8]. And, "ecosystem health" is peppered throughout the PSP Action Agenda. Thus, while we acknowledge the flaws and limitations of the phrase, we use it here because it is a familiar phrase that is salient in the policy arena.

The Future of Puget Sound: Where are We Going?

The charge is clear: restore the ecological health of Puget Sound by 2020. What is less clear, however, is what future the citizens of the Puget Sound region desire. Understanding what future we want, and what futures are possible, is critical to informing management decisions about complex systems such as Puget Sound, comprised of multiple unpredictable components. The theme of any individual vision of the future may range from particular ecosystem states (e.g., healthy orca populations, clean water) to socio-economic conditions (e.g., thriving ports, efficient and integrated public transportation). However, comprehensive visions of future states require that Puget Sound be considered in the context of a coupled social-ecological system, with the socio-economic system influencing the ecological system, and vice-versa. All components of this complex system are in turn being transformed by driving forces that can be either internal or external to the system. These unpredictable and largely uncontrollable driving forces, for example, climate change, the national and global economies, human desires, behavior and attitudes, each have their own potential trajectories that will help shape the future state of the Puget Sound ecosystem. For example, whether the future climate of Puget Sound is warmer and wetter, or warmer and drier, will certainly shape management strategies aimed at protecting species that use the freshwater streams and rivers in Puget Sound, such as salmon. Describing the future state of Puget Sound, therefore, goes beyond making predictions based on past observed trends in the ecological system and identifying actions that Puget Sound resource managers can implement. Understanding the myriad potential futures of Puget Sound is critical to setting targets aimed at achieving goals for restoring the health of Puget Sound by 2020.

This section will review previous efforts to describe alternate futures for Puget Sound, highlight the trade-offs inherent in these scenarios, particularly in light of drivers generated outside of the Puget Sound ecosystem, and draw connections between future scenarios and management strategies, including the importance of setting targets and deriving quantitative measures of progress. Finally, we suggest directions for continued efforts to describe alternate futures of Puget Sound.

1. Future States of Puget Sound

Describing the future state of Puget Sound has been approached in several ways, including using a formal scenario planning process, within the context of a regional planning strategy, using models and GIS (Geographical Information System) tools to map potential changes on the landscape, and setting specific targets for the desired future ecological system. Most of the work has been focused on the nearshore habitats of Puget Sound, with limited consideration of other domains of the ecosystem (e.g., rivers, forests, freshwater wetlands). Each approach described here is one component of what we see as a comprehensive future scenario process, beginning with a declaration of priorities by policy makers, followed by a thorough exploration of the driving forces behind the Puget Sound ecosystem and their potential trajectories, and finally, drawing explicit links (mediated by the driving forces) between potential policy decisions, biophysical states, and their consequences for the ecological system and ecosystem goals. As yet, there is no single "soup-to-nuts" approach to describing a future Puget Sound, though some of the efforts reviewed below are still works in progress.

Puget Sound Regional Council's Vision 2040

The Puget Sound Regional Council's "Vision 2040," adopted in 2008 and amended in 2009, is essentially a declaration of priorities for the future of Puget Sound by the major policymakers and politicians in the Central Puget Sound region [9]. Vision 2040 describes the growth management, environmental, economic and transportation strategies for the region. It co-prioritizes people, the economy, and the environment, and lists a series of goals and future actions, some of which are supported by existing policy. The document charts a pathway for land development and design, referencing existing land-use development policy (Washington State Growth Management Act) and establishes goals for matching development patterns with human well-being. Regional economic prosperity is a goal to be achieved by implementing a separately-established Regional Economic Strategy [10]. Finally, a multimodal regional transportation system is a priority, "integrating freight, ferries, highways, local roads, transit, bicycling and walking" [9].

Vision 2040 provides a framework within which regional planning on land use, economic development, and transportation can occur. The strategy explicitly takes into consideration the connectedness of regional planning and the environment. The document outlines goals, actions and implementation strategies for transportation and development, primarily from a policy and planning perspective. The drivers of the ecosystem are policies, which alter the (terrestrial) landscape according to a broad set of guidelines aimed at encouraging density within urban areas and limiting development outside of urban areas, and strengthening public transit and non-motorized transportation without compromising regional economic growth. There is a single vision of an ideal future Puget Sound region, and this document lays the groundwork for achieving that vision.

Summary: Within the context of a comprehensive effort to describe potential futures of Puget Sound, Vision 2040 serves as a statement by the citizens, as represented by their elected officials. Missing from this are more specific statements from the public about their views on, for example, a healthy Puget Sound. However, to date, no comprehensive survey or collection of citizen opinions about the future of Puget Sound exists, and therefore this document is the best proxy we have for gauging broad societal goals and desires. Any description of potential Puget Sound futures should include the public's desires as assurance that the ecosystem is headed in a direction supported by the public, and therefore this document is useful as one piece in the future scenario process.

Puget Sound Nearshore Partnership and University of Washington Urban Ecology Research Lab, "Future Scenarios

In another approach to describing a future Puget Sound, the Puget Sound Nearshore Partnership and the Urban Ecology Research Lab (UERL) produced "Future Scenarios" [11], which employs a formal scenario-building process to identify the driving forces of change in the Puget Sound ecosystem, and to develop multiple alternative scenarios based on the uncertainty in and interactions between those driving forces. Scenario building is a systematic method that has been applied to coupled social-ecological systems by, for example, the Millennium Ecosystem Assessment [12], and aims to generate more flexible approaches to EBM through the incorporation of uncertainty and multiple knowledge types. The fundamental premise is that the future is unknown, and that it is a function of several key factors that interact to create multiple potential future outcomes.

Through a series of visioning exercises with stakeholders and experts on the Puget Sound social-ecological system, two “key” drivers (climate and human behavior/perceptions) and nine “supporting” drivers (demography, development patterns, economy, governance, knowledge/information, natural hazards, public health, and technology/infrastructure) were identified, as were the interactions among them. The “key” drivers represent the most important and uncertain driving forces relevant to the issue, in this case the nearshore ecosystem of Puget Sound. Based on the potential trajectories of the key drivers and their interactions with the supporting drivers, six scenarios were developed. Narratives of each scenario described the prosperity, human attitudes, climate regime, development patterns, governance structure and demographics of a future Puget Sound, primarily as a function of the key drivers, climate and human behavior/perceptions and without drawing explicit links to component of the ecological system. Each narrative was rooted in a storyline, described by society’s worldview, human-nature relationships, and future outlooks (i.e. optimistic vs. pessimistic, or positive about human-nature relationships vs. hostile towards the environment).

The six scenarios spanned a broad range of social and climatic conditions, coupled with resulting effects on the ecological system. For example, in the “Collapse” scenario, climate change manifested as drier and warmer conditions in Puget Sound, and human behavior was self-interested and focused on the near-term. High levels of resource extraction and pollution caused harm to ecosystem function. Poor economic performance and increasing government expenditures led to fewer investments in infrastructure and public services, and eventual out-migration of the population. On the other end of the spectrum, the “Forward” scenario described a future with only limited climate change in Puget Sound and a cooperative social ethic, leading to a proactive approach to environmental issues and higher quality of life. There was increased population and economic growth. There was a greater understanding of the linkages between society and nature, leading to a stronger relationship between residents and their environment.

Summary: “Future Scenarios” gives a very thorough treatment to the socio-eco-political matrix within which the nearshore ecosystem (to which this analysis was limited) exists. Links are drawn between attitudes, economics, politics and climate, and alternative trajectories are explored for each--an important acknowledgment that there is great uncertainty involved in any vision of the future. This approach to fleshing out ecosystem drivers and their trajectories is critical in a comprehensive effort to describe the future of complex social-ecological systems like Puget Sound. The next step of this project is to explicitly link the drivers and scenarios to the ecological constituents and interactions.

Future Risk Assessment Project (FRAP) and Ecosystem Portfolio Model (EPM)

The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP) has developed several future scenarios of Puget Sound by coupling the Future Risk Assessment Project (FRAP), the creation of one set of land-use scenarios, with the Puget Sound Ecosystem Portfolio Model (EPM; [13]), a suite of models that evaluate the effects of land-use scenarios on nearshore ecosystems. The Puget Sound Nearshore Science Team and scientists from Oregon State University generated land-use scenarios based on three potential directions for land-use policy: status quo, where current trends continue forward; managed growth, which incorporates aggressive policies directing growth into urban areas; and unconstrained growth, which relaxes land-use regulation. Each scenario modulates several parameters governed by growth policy: population distribution, urban and rural development patterns, nearshore development pattern/intensity, and protection of open space. These scenarios were input to a GIS model, generating terrestrial maps of land use/land cover for Puget Sound [14].

The EPM models link land-use patterns generated by policy scenarios to ecosystem state, and therefore analyses can be directed towards specific goals. One such set of links was developed targeting human well-being, one of the six major goals of the Puget Sound Partnership. Using a list of human well-being indicators chosen in consultation with multiple expert groups, explicit connections are drawn between land-use patterns and metrics of human well-being using existing data and models. For example, each land-use scenario developed by FRAP results in some degree of shoreline modification, which is then linked to indicators of human well-being, one example of which is recreational beach use. A statistical model predicts the effects of land-use development on recreational beach use as a function of recreational visit data, demand (based on population density) and access (based on travel cost), each of which is affected by shoreline development.

Summary: The FRAP/EPM approach emphasizes connections between patterns on the landscape, generated through simple policy-driven scenarios, and specific ecosystem states that can be linked to a broader ecosystem or policy goal, in this case human well-being. In the context of a comprehensive future scenario process, this is a critical step that highlights the consequences of individual policy decisions, like land-use development, for ecosystem goals, in this case human well-being. This technique could also be used in conjunction with scenarios that generate ranges of responses by the social-ecological system. For example, to these same land-use policy scenarios could added climate change scenarios that will alter the way the ecological system responses to, for example, shoreline modification. Under warmer, wetter conditions, erosion patterns and the absolute amount of shoreline in Puget Sound may change, both of which will affect recreational beach use. This tool linking changes made on the landscape to ecosystem goals is helpful in charting a path towards ecosystem goals and in predicting the feedbacks of policy decisions.

Puget Sound Salmon Recovery Plan

The Puget Sound Salmon Recovery Plan, in contrast to the above approaches, uses specific targets to describe the future, by establishing regional and watershed-specific abundance and productivity targets for threatened Pacific salmon and bull trout populations. In 1999, Puget Sound Chinook Salmon, Coastal/Puget Sound bull trout and Hood Canal summer chum were listed as threatened under the Endangered Species Act (ESA). Subsequently, a number of independent recovery plans for Puget Sound salmon populations were initiated, and the Puget Sound Salmon Recovery Plan aimed to combine the efforts and strategies of several groups, most notably the Shared Strategy for Puget Sound (Shared Strategy) and NOAA's National Marine Fisheries Service [15]. The Shared Strategy generates individual watershed targets for salmon populations based on technical models and historic information, setting target ranges for salmon abundance and productivity.

Using these watershed-specific targets, the Salmon Recovery Plan then establishes short- and long-term numerical goals, identifies limiting factors, and offers specific strategies, in some cases at the scale of individual tributaries, for reaching those goals. For example, the Lake Washington/Cedar River/Lake Sammamish Chinook salmon population's 10-year goal is 1,600 spawners, and the long-term goal is between 2,000-12,000 spawners, allocated among the different water bodies. The major limitations to achieving increases in productivity and abundance include altered hydrology, loss of riparian vegetation, lack of woody debris, and high temperatures and pollution levels. The strategies identified to achieve the abundance and productivity goals include protecting and managing upper watersheds, restoring stream habitat, improving lake habitat and reducing the impacts of urban development. Individual actions are recommended for specific tributaries or water bodies.

The Shared Salmon Recovery Plan defines the future in terms of specific targets for the ecological system (salmon abundance and productivity), identifies threats to achieving those targets, and lays out strategies and actions for addressing the threats. While it does not offer alternate future scenarios, it outlines an adaptive management approach to investigate and incorporate sources of uncertainty such as climate change, interactions between wild and hatchery fish, effects of poor freshwater and marine water quality, and nearshore habitat processes.

Summary: This approach is one of few that specifically identifies targets for Puget Sound ecosystem goals. In the context of a complete results chain approach to achieving a healthy Puget Sound, setting targets is critical for understanding the trade-offs between different goals (see below). In the context of a comprehensive future scenario process for Puget Sound, targets represent concrete objectives against which results from statistical models (e.g., EPM) and potential future states of driving forces can be compared. For example, under a warmer, wetter climate, with a population focused on near-term objectives, a flat local economy and status-quo land use policies, can the stated salmon productivity targets be reached for each watershed? Under which scenarios are the targets achievable? Asking these complex questions highlights the need for a comprehensive effort to describe the future Puget Sound.

Summary of Future Scenario Efforts

The above review of four very distinct efforts to describe a future Puget Sound highlights what is needed, and what is missing, in a comprehensive future scenario process. Comprehensive visions of a future Puget Sound will chronicle the political motivation and citizens' desired state; explore the uncertainty in the driving forces of the social-ecological system, including climate change; draw explicit links between the drivers and the ecological state; and develop targets for future state characteristics based on existing data and models. "Vision 2040" provides the best measure we have of the public's vision for the future of Puget Sound; however, this description is missing specific references to the ecological system which could help management predict the public's response to or support for certain decisions or trade-offs. Characterizing the major uncertainties in the system and offering potential future scenarios based on these is a crucial step in adequately matching ecosystem goals with strategies and actions, and "Future Scenarios" is a very thorough treatment of the driving forces behind this uncertainty. Any thorough approach to describing potential futures must incorporate climate scenarios, as well as the key socio-economic drivers in the system. If these driving forces can be incorporated into the model-based scenarios and on-the-ground biophysical depictions of policy decisions (effectively exemplified by FRAP and EPM), then more accurate assessments of alternate management strategies will be possible. This is a formidable task, and the work reviewed above contributes towards that end. A thorough effort to describe a future Puget Sound (i.e., Where are we going?) is a partner to larger effort in this document, developing indicators for the system (Are we there yet?).

Key point: Characterizing the major uncertainties in the system and offering potential future scenarios based on these is a crucial step in adequately matching ecosystem goals with strategies and actions. Any thorough approach to describing potential futures must incorporate climate scenarios, as well as the key socio-economic drivers in the system.

Trade-offs and Targets

Among other marine ecosystem management programs in North America, the most common approach to defining the future is akin to the FRAP/EPM method described above: develop predictions for future ecological states based on existing information, and specifically, generate a few land-use scenarios based on policy decisions governing development, growth management, pollution controls, transportation and/or conservation, and connect the resulting landscape patterns to ecological function, such as nutrient or sediment inputs (e.g. [16, 17]). Less common is a thorough examination of the socioeconomic and climate drivers of ecosystem dynamics, as in the UERL/PSNERP "Future Scenarios." However, even in cases where the drivers of the ecosystem are well described and incorporated into future scenarios, their utility is limited by the extent to which linkages are drawn between drivers, ecological state, and goals or targets.

Most future scenario-building efforts (including several reviewed above), lack an explicit treatment of the trade-offs required to successfully arrive at a desired future state. Moving from citizen desires to ecosystem reality requires confronting trade-offs among multiple goals. For example, the U.S. Government's roadmap for restoring the Louisiana-Mississippi Coast Ecosystem acknowledges that stakeholders must "jointly evaluate trade-offs that will likely be necessary" to meet the multiple goals of ecosystem function, resilience, economics and climate adaptation [18]. Such trade-offs are cast in sharp relief when considering the tension between local economic prosperity, the global economy and water quality in Puget Sound. The Ports of Seattle and Tacoma together comprise the third busiest container port in the U.S. [19], and a large proportion of the Puget Sound regional economy relies on the import and export of goods through the ports. A growing demand for imports and exports through Puget Sound ports, generated by a flourishing global economy, could increase shipping traffic. The Ports of Seattle and Tacoma are already challenged to meet port productivity goals as well as water quality requirements, and a rise in traffic through the Ports would exacerbate that particular challenge, if not necessitate additional construction along Puget Sound shorelines. Both increased shipping traffic and increased hardening of shorelines negatively impact Puget Sound marine species, food webs, habitat, water quality – each a PSP goal. Other trade-offs likely to emerge include those between population increase, development pressures and habitat protection; population increase, agricultural demands and minimum stream flows; and economic prosperity, shipping traffic and invasive species control. As these examples highlight, achieving human well-being and ecological function without sacrificing economic prosperity in Puget Sound will require some compromises.

In some cases, thorough consideration of trade-offs is not possible owing to the absence of targets--the desired future numeric value for an ecosystem indicator. In large part, quantifiable targets related to the state of the Puget Sound ecosystem are missing from future scenario efforts (one major exception to this is the Shared Salmon Recovery Plan). In the absence of targets, the assessment of progress and a complete understanding of trade-offs are elusive. Establishing targets forces confrontation with trade-offs; without targets, the definition of "success" – and the route to get there – is flexible. Furthermore, in the context of a future scenario process, evaluation of scenarios is hampered without targets. Full evaluation of trade-offs, in turn, involves describing the human drivers of ecosystem change, such as behavior and perception, which highlights the importance of including these driving forces in future scenario processes.

Key point: Establishing ecosystem targets is essential as it forces confrontation with trade-offs among targets. Full evaluation of trade-offs requires examination of the human drivers and these driving forces should be central in future scenario processes.

Management Strategy Evaluation

One means of addressing trade-offs and targets is management strategy evaluation (MSE), a conceptual framework that facilitates testing and comparison of different management strategies designed to achieve specified management goals [20]. The MSE process is analogous in many ways to the approach employed by the FRAP/EPM effort described previously. Born from the concepts of adaptive management of resources [21] and management procedure evaluation [22], MSE is an analytical process that follows six basic steps:

- Policy objectives, target values, and performance measures (measures of success) for important resources are defined and quantified.
- A management strategy is designed to achieve the objectives.
- The strategy is implemented in an operating model that simulates ecosystem processes relevant to the resources of interest. The model may be simple or complex, depending on the underlying questions.
- A simulated monitoring program draws imperfect data from the operating model.
- An assessment model is run to determine the effect of management on indicator variables measured by the simulated monitoring program. The levels of the indicators are compared to the pre-determined target values; the difference is a measure of performance.
- Depending on the outcome of the assessment, decision rules will be activated that either continue or adjust the management strategy, until the objective is met.

This process is repeated for multiple management strategy alternatives, which allows comparison of different strategies—in terms of both successes (positive performance measures; rapid progress) and weaknesses (negative performance measures, slow progress)—in attaining desirable future states. In this way, the potential effectiveness and the potential trade-offs of the strategies are understood.

Several operating models that are available or in development could support MSE of alternate Puget Sound futures. Some available models focus on aquatic and marine issues such as municipal water supply [23] and the relationship between terrestrial activities and marine biogeochemistry (e.g., [24]). Others focus on terrestrial issues such as land use and urbanization impacts on species diversity [25]. Several models in development simulate the structure of the marine food web (e.g., the Ecopath with Ecosim model of Central Puget Sound [26]), and are well-suited to forecast trade-offs between different resources or stakeholders as a result of simulated management actions. Continued development of such models is a high priority.

Key Point: Formal Management Strategy Evaluation (MSE) is an important tool for assessing management scenarios. Several computer models are available that could support MSE, but continued model development should be a high priority.

An Expanded Results Chain Model

Future scenarios are a critical tool for informing and refining conservation strategies. The PSP has adopted the Open Standards for Practice of Conservation framework for performance management. A key component of the Open Standards is "results chains," which map management strategies to their expected outcome (e.g., reduction of a threat) and their impact on key components of the ecosystem (Figure 1). An individual results chain is comprised of multiple components: a goal is linked to a strategy, such as a policy decision, for achieving that goal; associated with each strategy are one or more outcomes of that strategy; a second outcome or set of outcomes describes an expected change in the ecosystem threat; the threat outcome is linked to an ecological impact, which relates to the goal (Figure 1). In the context of the Open Standards, alternate future scenarios, whether describing possible trajectories of external drivers (e.g., climate change, human attitudes), policy outcomes (e.g., Shoreline Management Act amendments), or the state of the economy, can be incorporated into results chains by generating ranges for outcomes or impacts, rather than single values. In this way, alternate futures help set realistic targets for desired ecological states.

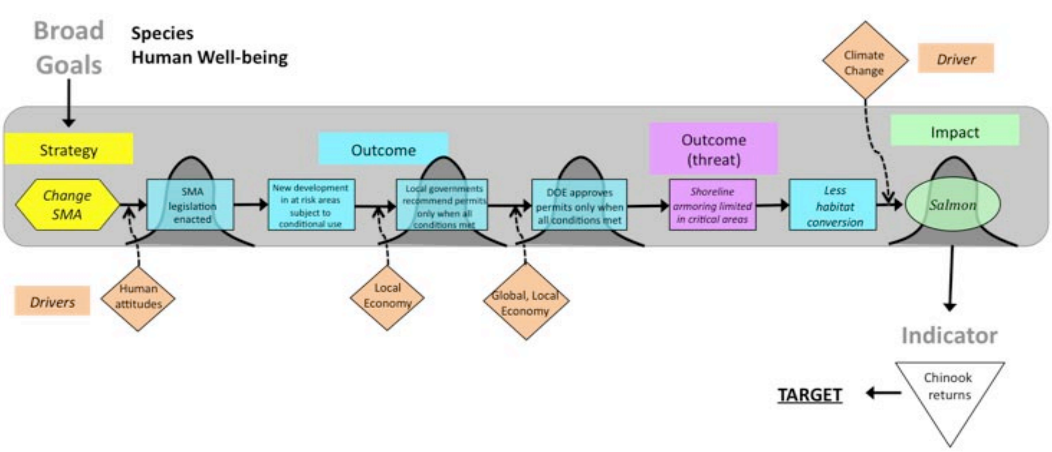


Figure 1. An example of a modified results chain, incorporating the influence of future scenarios of drivers (orange diamonds) on links in the chain, adding an example of an indicator (blue triangle) and showing where a target would be included. The effect of future scenarios on a results chain is shown here by overlaying a distribution of possible conditions (grey curves) for outcomes or impacts where they are potentially influences by future conditions of external drivers. Original chain from [4].

To illustrate the utility of future scenarios in the results chain framework, we use an example where a set of land protection actions from the Puget Sound Partnership's Action Agenda is aggregated into a results chain describing regulatory strategies for protecting and enhancing ecosystem components. One sub-chain focuses on a strategy to amend the Shoreline Management Act (SMA) by requiring conditional use permits for land development (Figure 1), with the ultimate objective of converting less habitat, which would positively impact many components of the ecosystem, including salmon [4]. The first "if...then" step in this sub-chain is that if the SMA is amended, then the revised version will be enacted. This initial step requires approval by voters, through their elected legislative representatives, and is therefore subject to the influence of human attitudes and perceptions. Surveys of Puget Sound citizens and stakeholders have indicated that, in general, people do not think Puget Sound is alarmingly unhealthy, and they are disinclined to make major sacrifices to protect and restore the ecosystem [27]. Therefore, there is some uncertainty, a function of human attitudes, about whether this legislation would be approved, and that uncertainty is described by a range of potential policy outcomes, rather than a single deterministic outcome. In addition, assuming all the outcomes in the results chain are achieved, and less habitat is converted by development, climate change can still influence the abundance and productivity of salmon populations through other mechanisms, and the impact of regulation changes on salmon will be mediated by the potential influence of climate change. Therefore, the goal "Salmon" is represented as a range of possible salmon populations, rather than a single value. This example illustrates the role of future scenarios in developing performance measures and outcomes for conservation plans.

We have also modified the results chain by adding in indicators, which are connected to the Impact (Goal) – in this case, the indicator of "Salmon" is "Chinook returns." Associated with each indicator, also, would be a target, in this case, likely watershed-specific targets for Chinook salmon returns, such as those generated by the Shared Salmon Strategy.

"The future ain't what it used to be." Y. Berra

Our review of the few efforts to envision a future Puget Sound suggests considerable room for future work. While there is clear agreement that the future state of Puget Sound should be different than it is now, the region lacks a lucid vision of the desired state of the coupled human-ecological system. The strong links between human activities and nearshore ecosystem components have resulted in most of the effort being direct towards this domain; however, there is no doubt that future scenarios for the whole of Puget Sound - from "sea to summit"- are required. Externalities of human and natural origin are important driving forces in this coupled system and should be included in analyses of scenarios. And, ultimately, these scenarios are most useful if they identify trade-offs and develop means for operating along the axes between trade-offs. The lack of management targets for most components of the Puget Sound ecosystem allows managers and policy makers to avoid confronting many trade-offs and thus encourages somewhat narrow (e.g., single ecosystem domains) or vague and ill-defined visions of the future. However, our review reveals that the foundation to generate scenarios of a future Puget Sound is in place. As the efforts described here continue and expand and new endeavors begin, we expect more comprehensive visions of Puget Sound's possible future to emerge.

Key point: While there is clear agreement that the future state of Puget Sound should be different than it is now, the region lacks a lucid vision of the desired state of the coupled human-ecological system. However, the foundation to generate scenarios of a future Puget Sound is in place. As the efforts described here continue and expand, we expect more comprehensive visions of Puget Sound's possible future to emerge.

Table 1. Summary of final scenarios generated by "Future Scenarios"; adapted from Table 6.1 in [11].

Forward: Low climate change coupled with a greater social ethic of cooperation provided the Puget Sound the opportunity and resources to proactively address environmental problems and improve the quality of life for all of its residents. While the region's economy continued to grow and immigration doubled the Sound's population, the region managed to maintain and restore ecological function. Residents, governments and industry shared a new understanding of the Puget Sound ecosystem as an integrated human-ecological system creating a renewed relationship with their environment.
Order: While climate change was a best-case scenario, population growth coupled with increasing consumption placed pressure on the Puget Sound's resources. An increasingly fragmented governmental structure spurred conflict between municipalities and interest groups. In spite of existing environmental regulations, a lack of coordination among governmental agencies was a major obstacle in improving ecosystem function. Sprawling developments coupled with a low investment in the region's infrastructure, education and health significantly reduced the quality of life in the region.
Innovation: More and greater climate fluctuations increased the Puget Sound's vulnerability to floods, windstorms and fires. Technological innovation mitigated negative impacts on residents and infrastructure. The high tech industry led the regional economy, drawing in skilled labor and high wages and largely controlling the political arena. Growth rates of new ideas, production, immigration and housing development all increased, generating wealth and jobs. Innovation allowed per capita consumption levels to remain high through increased efficiency and closed-loop industrial processes.
Barriers: Society in the Puget Sound region divided as the disparity between the rich and poor was magnified. Escalating climate impacts posed significant threats to private property, regional infrastructure and natural resources. Residents responded by building stronger walls, moving uphill and securing their investments. As cost of fuel and mitigation rose, the rich buffered their families from impeding harm, while the poor were left behind with a continuously degrading economy. Government regulations were relaxed in an effort to overcome financial hardships, but instead facilitated a growing economic divide and poor management decisions.
Collapse: Decreased precipitation rates, warmer temperatures and a self-interested short term

society spelled disaster for the Puget Sound region. Resource extraction and pollution load exceeded critical thresholds causing harm to ecosystem functions. Increased fragmentation and decreased precipitation led to droughts, forest fires and massive pest outbreaks. Increasing government costs and dwindling resources led to poor investments in infrastructure improvements and public services. As the beauty and health of the Puget Sound landscape slipped so did major industries, causing a severe economic depression followed by out-migration.
Adaptation: Despite major challenges caused by climate change, adaptive management and a positive consciousness regarding environmental change allowed the region to cope with the emerging problems and maintain high standards of life. Cooperation among residents, businesses and governmental units allowed this region to prosper despite increased vulnerability brought on by climatic impacts. Production rates decrease, but collective wealth rose due to investment is education, health and shared community resources such as public transit and renewable resource infrastructure. A growing awareness of future uncertainty embedded the precautionary principle into resource management and environmental policies, erring on the side of caution and increasing the region's resiliency.

An Approach to Selecting Ecosystem Indicators for Puget Sound

1. Background

What are ecosystem indicators and why are they useful?

Ecosystem indicators are quantitative biological, chemical, physical, social, or economic measurements that serve as proxies of the conditions of attributes of natural and socio-economic systems [28-31]. Ecosystem attributes are characteristics that define the structure, composition and function of the ecosystem that are of scientific and/or management importance, but insufficiently specific and/or logistically challenging to measure directly [28-31]. Thus, indicators provide a practical means to judge changes in ecosystem attributes related to the achievement of management objectives. They can also be used for predicting ecosystem change and assessing risk.

Terminology and Concepts

Indicators	Quantitative biological, chemical, physical, social, or economic measurements that serve as proxies of the conditions of attributes of natural and socioeconomic systems.
Key Attributes	Characteristics that define the structure, composition, and function of a Focal Component.
Focal Components	Major ecological characteristics of an ecosystem.
Goals	Combine societal values and scientific understanding to define a desired ecosystem condition.
DPSIR framework	Driver-Pressure-State-Impact-Response (DPSIR). Drivers are factors that result in pressures that cause changes in the system. Pressures are factors that cause changes in state or condition. State variables describe the condition of the ecosystem. Impacts measure the effect of changes in state variables. Responses are the actions taken in response to predicted impacts.

For more information and links to references, see Glossary

Ecosystem indicators are often cast in the Driver-Pressure-State-Impact-Response (DPSIR) framework—an approach that has been used by the PSP and broadly applied in environmental assessments of both terrestrial and aquatic ecosystems, including NOAA's Integrated Ecosystem Assessment [32]. *Drivers* are factors that result in pressures that cause changes in the system. Both natural and anthropogenic forcing factors are considered; an example of the former is climate conditions while the latter include human population size in the coastal zone and associated coastal development, the desire for recreational opportunities, etc. In principle, human driving forces can be assessed and controlled. Natural environmental changes cannot be controlled but must be accounted for in management. *Pressures* are factors that cause changes in state or condition. They can be mapped to specific drivers. Examples include coastal pollution, habitat loss and degradation, and fishing. Coastal development results in increased coastal armoring and the degradation of associated nearshore habitat. *State* variables describe the condition of the ecosystem (including physical, chemical, and biotic factors). Impacts comprise measures of the effect of change in these state variables such as loss of biodiversity, declines in productivity and yield, etc. *Impacts* are measured with respect to management objectives and the risks associated with exceeding or returning to below these targets and limits. *Responses* are the actions (regulatory and otherwise) that are taken in response to predicted impacts. Forcing factors under human control trigger management responses when target values are not met as indicated by risk assessments. Natural drivers may require adaptational response to minimize risk. For example, changes in climate conditions that in turn affect the basic productivity characteristics of a system may require changes in ecosystem reference points that reflect the shifting environmental states.

Ideally, indicators should be identified for each step of the DPSIR framework such that the full portfolio of indicators can be used to assess ecosystem condition as well as the processes and mechanisms that drive ecosystem health. State and impact indicators are preferable for identifying the seriousness of an environmental problem but pressure and response indicators are needed to know how best to control the problem [33]. However, because of time constraints, we opted to focus this initial draft of the PSSU on indicators of ecosystem state. Of course, the distinctions between pressure, state, and impact are often muddled and depend very much on perspective. For example, water quality is a primary goal of the PSP, and thus indicators of water quality provide information on the state of this goal. However, poor water quality is clearly a pressure that affects other states (e.g. species and food webs) and impacts (e.g. recreational fisheries). Thus, although we do not focus on driver, pressure and impact indicators, many are included in this section as well as the section on indicators of human health and well-being. It is also important to note that Chapters 1 and 2 of the PSSU are using indicators as tools to assess ecosystem status and condition, while Chapter 3 will focus on drivers and pressures of change to Puget Sound.

Relationship to previous indicator work in Puget Sound

The development of indicators for the Puget Sound ecosystem has a long history with different groups adopting slightly different frameworks to meet their varying goals [1, 34-40]. Here, we build upon the history of indicator work in the region, extending and adopting it to the current management setting in Puget Sound. We accomplish this in several ways. First, we propose a framework that links indicators to both PSP ecosystem recovery goals and the PSP performance management system. Additionally, we embrace and expand the criteria for indicator selection suggested by O'Neill et al. (2008) as part of their earlier indicator vetting for the PSP [34]. We also extend previous evaluations by considering potential indicators for which data are currently unavailable but are otherwise deserving of attention. Finally, while previous evaluations emphasized expert opinion, our approach focuses on peer-reviewed literature, supplemented by other sources of information.

In the 2008 Action Agenda, the PSP articulated six outcome statements that defined key attributes corresponding to each of the PSP ecosystem recovery goals [1]:

- Human health is supported by clean air and water, and marine waters and freshwaters that are safe to come in contact with. In a healthy ecosystem the fish and shellfish are plentiful and safe to eat, air is healthy to breathe, freshwater is clean for drinking, and water and beaches are clean for swimming and fishing.
- Human well-being means that people are able to use and enjoy the lands and waters of Puget Sound. A healthy ecosystem provides aesthetic values, opportunities for recreation, and access for the enjoyment of Puget Sound. Tribal cultures depend on the ability to exercise treaty rights to fish, gather plants, and hunt for subsistence, cultural, spiritual, ceremonial, and medicinal needs. The economic health of tribal communities depends on their ability to earn a livelihood from the harvest of fish and shellfish. Human well-being is also tied to economic prosperity. A healthy ecosystem supports thriving natural resource and marine industrial uses such as agriculture, aquaculture, fisheries, forestry, and tourism.
- Species are "viable" in a healthy ecosystem, meaning they are abundant, diverse, and likely to persist into the future. Harvest that is consistent with ecosystem conditions and is balanced with the needs of competing species is more likely to be sustainable. When ecosystems are healthy, non-native species do not impact the viability of native species or impair the complex functions of Puget Sound food webs.
- Marine, nearshore, freshwater, and terrestrial habitats in Puget Sound are varied and dynamic. The constant shifting of water, tides, river systems, soil movement, and climate form and sustain the many types of habitat that nourish diverse species and food webs. Human stewardship can help habitat flourish, or disrupt the processes that help to build it. A healthy ecosystem retains plentiful and productive habitat that is linked together to support the rich diversity of species and food webs in Puget Sound.
- Clean and abundant water is essential for all other goals affecting ecosystem health. Freshwater supports human health, use, and enjoyment. Instream flows directly support individual species and food webs, and the habitats on which they depend. Human well-being also depends on the control of flood hazards to avoid harm to people, homes, businesses, and transportation.
- Water quality in a healthy ecosystem should sustain the many species of plants, animals, and people that reside there, while not causing harm to the function of the ecosystem. This means pollution does not reach harmful levels in marine waters, sediments, or fresh waters.

In order to evaluate the status and condition of the ecosystem and progress towards recovery, it is necessary to have a more specific and structured list of attributes that define the characteristics of the ecosystem, as well as identify potential indicators for these attributes. Clearly, there is no shortage of potential indicators. However, an enormous challenge lies in winnowing down the catalog of candidate indicators to a manageable list that are most likely to faithfully track all of the important attributes of ecosystem health and, in so doing, enables further progress toward the PSP goals.

Our approach to selecting and evaluating a suite of indicators for the Puget Sound ecosystem was to: 1) develop a framework to describe the key ecosystem attributes of Puget Sound, organized by each of the PSP goals (Section 3.2), 2) select and organize potential environmental indicators according to the key ecosystem attributes (Section 3.2.3-3.24), 3) select a set of criteria to evaluate individual indicators (Section 4), and 4) evaluate the individual indicators according to a set of explicit criteria (Section 5) (see [41]). These steps will be described below.

A framework for selecting indicators within the management context of Puget Sound

Selecting a suite of indicators that accurately characterize the ecosystem, while also being relevant to policy concerns, is a significant challenge. A straightforward approach to overcoming this challenge is to employ a framework that explicitly links indicators to policy goals [42, 43]. This type of framework organizes indicators into logical and meaningful ways in order to assess progress towards policy goals. For example, Niemeijer and de Groot (2008) show that in the absence of an organizing framework, different indicators can be selected for the same environmental issue, even when evaluation criteria and data availability are similar [33]. Without a clearly defined link between the environmental issue (or policy goal) and the list of indicators, it becomes impossible to tell which set of indicators best characterizes the issue and why. Ideally, each indicator has a particular function or role in evaluating the status of an environmental concern. A well-defined and transparent framework clearly demonstrates why particular indicators were chosen (i.e., what function is fulfilled by each indicator), why others were ignored, and how the chosen set of indicators best address the environmental issue. Thus a framework is crucial for placing environmental issues and indicators into context so that indicators are selected based on analytical logic rather than individual indicator characteristics [33]. It also helps avoid redundancies and identifies gaps where indicators are needed.

In the 2008 Action Agenda, the PSP discussed the need for an organizing framework to analyze ecosystem information and provide an integrated assessment of the status of Puget Sound [1]. Several frameworks have since been developed by the Partnership, however no framework has been formally adopted [37]. Previous frameworks were developed based on general recommendations and guidance in the Open Standards for the Practice of Conservation, and reports by the U.S. EPA, and the Heinz Center [3, 42, 44]. We have drawn upon these documents, as well as Harwell et al. (1999), to develop a broad, hierarchical framework to guide our evaluation of Puget Sound ecosystem indicators [43].

A guiding principle in the development our framework was that it should be reflective of societal goals and values, and be policy-relevant [3, 41-43]. The clearest guidance available for values and policy relevance are the six statutory goals defined by the PSP. Our framework thus begins with these six Goals. We then decompose these goals into unique ecological Focal Components within specific habitat domains (i.e., marine, freshwater, terrestrial, and interface/ecotone). Each focal component is characterized by Key Attributes, which describe fundamental aspects of each focal component. Finally, we map Indicators onto each ecosystem key attribute (Figure 2). Each tier of this framework is detailed below.

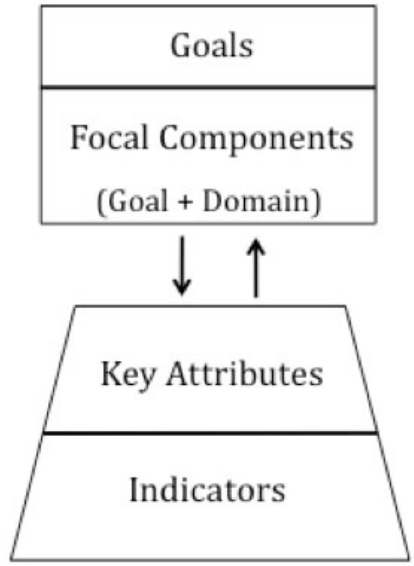


Figure 2. Proposed framework organization for assessing and reporting on ecosystem condition in Puget Sound.

Tier 1: Goals.

The broadest category of division of our framework is Goals. Goals combine societal values and scientific understanding to define a desired ecosystem condition [42, 43]. Explicit descriptions of the societal values related to the condition of Puget Sound are encompassed in the six statutory goals developed by the PSP [37], as shown in Section 3.1.3.

These goals reflect both societal and ecological interests in Puget Sound, and have been used as the fundamental organizing framework for assessing a 'healthy' Puget Sound ecosystem in the Partnership Action Agenda [37]. They are policy-relevant, which is foundational in the development of this framework. Note that for the purposes of indicator evaluation, we separated "Species" and "Food Webs." This section focuses only on natural ecosystem components. Thus, human health and human well-being are addressed elsewhere in the PSSU.

Tier 2: Focal Components.

Focal Components are the major ecological characteristics of an ecosystem that can be used to organize relevant information in a limited number of discrete, but not necessarily independent categories [3]. In the Open Standards for the Practice of Conservation they are referred to as, 'focal conservation targets.' The term 'Focal Component' has been used previously by the PSP [37] and has been adopted here to keep terminology consistent.

Focal Components were derived by dividing each of the Goals into distinct habitat domains that are characterized by unique qualities or traits. The domains we chose were marine, freshwater, terrestrial, and interface/ecotone. The interface/ecotone domain includes zones with a combination of traits from the other major groups such as the nearshore environment, wetlands, and estuaries.

This grouping (Table 2) provides a comprehensive view of the major ecological characteristics of Puget Sound based on area, and allows Focal Components to be assessed at an individual level (e.g., marine habitats), or aggregated into a single environment (e.g., assessing the integrity of the marine environment across all marine-related Focal Components).

Table 2. Summary of Focal Components based on goal and domain.

Goal	Domain	Focal Component
Species	Marine	Marine Species
	Freshwater	Freshwater Species
	Terrestrial	Terrestrial Species
	Interface/Ecotone	Interface Species
Food Webs	Marine	Marine Food Webs
	Freshwater	Freshwater Food Webs
	Terrestrial	Terrestrial Food Webs
	Interface/Ecotone	Interface Food Webs
Habitats	Marine	Marine Habitats
	Freshwater	Freshwater Habitats
	Terrestrial	Terrestrial Habitats
	Interface/Ecotone	Interface Habitats
Water Quality	Marine	Marine Water Quality
	Freshwater	Freshwater Quality
	Interface/Ecotone	Interface Water Quality
Water Quantity	Freshwater	Freshwater Quantity

Tier 3: Key Attributes.

Key Attributes are ecological characteristics that specifically describe the state of Focal Components. They are characteristic of the health and functioning of a focal component. They are explicitly defined based on each Focal Component and provide a clear and direct link between the Indicators and Focal Components. A similar tier has been identified by the PSP and others. A part of our framework development was an explicit comparison of the Key Attributes developed here with those suggested in the other reports. Although they differ in detail, the Key Attributes adopted here encompass all those identified by the EPA (2002), Heinz Center (2008), and the PSP [37, 42, 44]. Selected Key Attributes are shown in Table 3.

Table 3. Selected key attributes for each goal. Definitions (or measures) are meant to describe what is meant by each attribute. For example, population size is represented by the number of individuals in a population or the total biomass.

Goal	Key Attribute	Relevant Measures
Species	Population Size	Number of individuals or total biomass; Population dynamics
	Population Condition	Measures of population or organism condition including: Age structure; Population structure; Phenotypic diversity; Genetic diversity; Organism condition
Food Webs	Community Composition	Species diversity; Trophic diversity; Functional redundancy; Response diversity
	Energy and material flow	Primary production; Nutrient flow/cycling
Habitats	Habitat Area & Pattern/Structure	Area or extent; Measures of pattern/structure including: Number of habitat types; Number of patches of each habitat; Fractal dimension; Connectivity
	Habitat Condition	Abiotic & biotic properties of a habitat; Dynamic structural characteristics; Water & benthic condition
Water Quality	Hydrodynamics	Measures such as: Water movement; Vertical mixing; Stratification; Hydraulic residence time; Replacement time
	Physical/Chemical Parameters (Sediments & Water Column)	Measures such as: Nutrients; pH; Dissolved oxygen/redox potential; Salinity; Temperature
	Trace Inorganic & Organic Chemicals (Sediments & Water Column)	Measures such as: Toxic contaminants; Metals; Other trace elements & organic compounds
Water Quantity	Surface Water	Hydrologic Regime Measures such as: Flow magnitude & variability; Flood regime; Stormwater
	Groundwater Levels & Flow	Groundwater accretion to surface waters; Within groundwater flow rates & direction; Net recharge or withdrawals; Depth to groundwater
	Consumptive Water Use & Supply	Water storage

We reduced the list of potential attributes for each Goal and Focal Component to two or three Key Attributes for two reasons. First, this approach is driven by a need for simplicity, succinctness, and transparency in the development of an organizing framework. Second, the use of only 2-3 attributes for each Goal and Focal Component provides a means to address data gaps in the selection and evaluation of indicators. By defining the key attributes broadly, our framework allows for situations in which a single attribute (e.g., population condition for the Species Goal) can be informed by multiple types of indicators depending on information availability (e.g., population condition can be tracked using data on disease for some species, data on age structure for others, etc.).

A discussion of the Key Attributes for each goal follows.

Key Attributes – Species

A central goal identified by the PSP is to have 'healthy and sustaining populations of native species in Puget Sound' that provide ecosystem goods and services to humans, and support the structure and functioning of the ecosystem itself [1]. Many different attributes can describe whether a population is 'healthy and sustaining'. For example, the U.S. EPA (2002) identified eight different measures (i.e., attributes) of species condition including population size, genetic diversity, population structure, population dynamics, habitat suitability, physiological status, symptoms of disease or trauma, and signs of disease [42]. Similar attributes identified by Fulton et al. (2005) included biomass, diversity, size structure, and spatial structure [45]. Niemi and McDonald (2004) suggest attributes based on type, for example, structural attributes include genetic structure and population structure whereas functional attributes include life history, demographic processes, genetic processes, and behavior [46].

Historically the PSP has focused on population size as the species attribute, recognizing that species health or condition was encompassed by most other PSP goals [40]. More recently the PSP identified species key attributes by applying the Open Standards to the Action Agenda [37]. The species attributes they selected were forage fish, condition of key fish populations, population size and condition of key marine shellfish and invertebrates, population size and condition of key marine mammals, population size and condition of key marine birds, extent of all salmon species, condition of all listed salmon species, spatial structure of all listed salmon species, and population size and condition of key terrestrial bird species [37].

Population size is defined as the number of individuals in a population or the total biomass of the population. Population dynamics that influence changes in abundance over time are also included. Population condition combines several measures: population structure, age structure, genetic diversity, phenotypic diversity, and organism condition.

Selection of Species Attributes in Puget Sound

Ecological attributes are intended to describe the state of an ecological system; in the case of species attributes, they are meant to describe the condition or viability of populations of species in an area. Measures of population condition or viability are important indicators, yet monitoring the status of all species is practically impossible. To address this, focus should be placed on identifying species indicators that characterize key interests in the region (i.e., focal species). For example, some species exert a disproportionately important influence on ecosystem condition, while others relate to biodiversity or are of direct interest to society. Examples of focal species include target, charismatic, vulnerable, and strongly interacting species. Target species are those fished or harvested for commercial gain or subsistence. Flagship species are those with widespread public appeal that are often used to communicate to the public about the condition of the ecosystem. Vulnerable species are those recognized with respect to their conservation status, for example, threatened, endangered, or of greatest conservation concern. Strongly interacting species (e.g., keystone species) are those whose presence, absence or rarity leads to significant changes in some feature of the ecosystem (adapted from [47, 48]).

The following sections provide examples of the utility of population size and population condition in evaluating the status of focal species as well as ecosystem health.

Population size

Monitoring population size, in terms of total number of individuals or total biomass, is important for management and societal interests. For example, abundance estimates are used to track the status of threatened and endangered species and help determine whether a species is recovering or declining. Accurate estimates of population biomass of targeted fisheries species are used to assess stock viability and determine the number of fish that can be sustainably harvested from a region. While population size can be used to assess population viability, more accurate predictions of viability can be obtained by including the mechanisms responsible for the dynamics of the population. Population dynamics thus provide a predictive framework to evaluate the combined effect of multiple mechanisms of population regulation (e.g., birth and death rates, immigration and emigration) to evaluate changes in abundance through time.

Population condition

Whereas the preceding attribute is concerned with measures of population size, there are instances when the "health" of the population may be of interest. For example, monitoring changes in population condition may presage an effect on population size or provide insight into long-term population viability. The dynamics of many populations are better understood through knowledge of population condition such as organism condition, age structure, genetic diversity, phenotypic diversity, and population structure. Impaired condition of any or all of these subcategories indicates biological resources at risk. In addition, monitoring changes in organism condition can be used to infer changes in environmental conditions.

Organism condition

Organism condition represents both physiological and disease status. Monitoring organism condition may help predict changes in population size, and reveal environmental problems that warrant management action. Past efforts by the PSP have focused on organism condition (e.g., toxins in harbor seals) as an indicator of Water Quality. While this may be applicable for organisms at lower trophic levels (i.e., because they respond at shorter temporal scales), but time lags associated with the transfer of toxins through the food web means that higher trophic level organisms (e.g., killer whales, sixgill sharks) are unlikely to reveal Water Quality issues at time scales relevant to management. We suggest these measures (e.g., toxins in killer whales) are better served as an indicator of species population condition.

Physiological status is the key mechanism linking both organism and population to their environment [49]. For example, individuals experiencing increased environmental stress may increase levels of stress hormones, eventually killing the individuals and leading to a decrease in population size. In the Galapagos, marine iguanas increased stress hormone levels due to fouling from an oil spill. The increase in stress hormone levels predicted a decrease in survival by approximately fifty percent, which was later confirmed by field studies [50]. Disease status can affect population size and dynamics as well. In Prince William Sound, viral hemorrhagic septicemia virus (VHSV) was linked to a reduction in Pacific herring recruitment [51]. A recent paper by Landis and Bryant (2010) suggests that disease prevalence in Puget Sound was a contributing factor to the decline of Pacific herring (Cherry Point, Squaxin Pass, Discovery Bay, and Port Gamble stocks) in the 1970s and 1980s [52]. Thus, monitoring organism condition may signal declines in population abundance before it occurs.

Monitoring organism condition is particularly important for long-lived organisms (e.g., marine mammals, rockfish) that live in contaminated habitats. Declines in population size of long-lived species may be slow to appear because of their long cohort turnover times. The temporal scale at which this occurs makes it difficult to recognize the population is in decline, and respond fast enough to prevent severe changes in population dynamics [53]. Declining organism condition from contaminant exposure can also interact with diseases so that individuals in poor physiological condition are more susceptible to infections [54]. In juvenile salmon, exposure to contaminants lead to increased disease susceptibility, significantly reducing population size [55].

Finally, examining the physical condition of a population may reveal problems with current management strategies. For example, salmon injured by gillnets show reduced survival and fail to reproduce; this suggests estimates of spawning stocks, which count injured fish as part of the aggregate escapement of viable spawners, are inflated [56].

The remaining subcategories of population condition (i.e., age structure, population structure, genetic diversity, and phenotypic diversity) are primarily used for assessing focal species condition, and generally do not present information relating to environmental conditions. Due to this reason, these subcategories are discussed in terms of relevance to focal species.

Age structure

Population age structure is used to estimate population viability by modeling population trends through time, and can be especially useful for evaluating the long-term stability of a population. Monitoring age structure may also be useful in attributing declines in abundance to specific factors, which may otherwise be difficult to detect.

Robust age structure (i.e., multiple reproductive age classes) is critical for fish populations to withstand environmental variability and maintain resilience. Multiple reproductive age classes provide resilience for several reasons: (1) overall reproductive output increases, (2) age-related differences in spawning locations and timing allocate reproductive outputs across larger spatial and temporal areas, and (3) there is increased quantity and quality of eggs produced by older fish [57, 58]. Fisheries often target large and therefore old individuals, effectively truncating the age structure of the population. This is likely to reduce population resilience.

In order to attribute declines in stellar sea lion (SSL) populations to specific factors, age-structure information is required to separate out vital rate changes from population abundance estimates [59]. For example, a risk factor (e.g., contaminants) may affect an age-specific vital rate but show no corresponding change in population abundance. Examining age-structure trends may provide insight into population declines of various species in Puget Sound (e.g., Southern

Resident Killer Whales, Pacific herring, rockfish) or elucidate factors that affect age-specific organism condition.

Genetic diversity

Genetic diversity measures may be important in assessing long-term population viability, as well as the ability for a population to adapt to changing environmental conditions. Monitoring genetic loci or gene expression may also help detect the onset of selection events such as emerging diseases, climate change or land use change, or pollution [60].

Although not always the case [61], loss of genetic variation can reduce individual fitness (e.g., through loss of heterozygosity), as well as the ability of populations to evolve in the future (e.g., through loss of allelic diversity) [62]. For example, in Greater Prairie Chickens loss of genetic variation was linked with lower hatching success of eggs following population declines [63]. Genetic changes (e.g., declines in fecundity, egg volume, larval size, etc.) caused by overharvesting fish populations can increase extinction risks and reduce the capacity for population recovery [64].

Phenotypic diversity

Individual organisms adapt to changing environmental conditions by sensing the changes and responding appropriately, for instance, by switching their behavior or physiology. However this means that every individual must reserve a portion of their energy to actively sensing and adapting to environmental changes. An alternative strategy is to diversify a population: each subset of the total population is adapted to a slightly different environmental condition (i.e., phenotypic diversity). Sockeye salmon, for example, show a suite of adaptations to the diversity of spawning habitats. This phenotypic diversity has proven to be critical under changing environmental conditions in Bristol Bay, Alaska. As conditions changed, populations demonstrated differential responses so that at different times, different populations became more productive [65]. In California, the development of the Sacramento-San Joaquin watershed has truncated the life history diversity of Chinook salmon, resulting in the collapse of these populations [66]. Recognizing and understanding phenotypic diversity may prevent the loss of population subsets that currently appear unproductive, but may prove vital for long-term population sustainability.

Population structure

Population structure refers to spatial dynamics, or how different populations interact in space. In many instances local populations are linked, thereby creating a metapopulation. When environmental conditions change, some populations decline while others persist, but the overall density of the metapopulation may remain relatively steady. Metapopulations persist through a suite of adaptations at the individual (e.g., physiological and behavioral adaptations) and population level (e.g., each subpopulation lives in a separate location and contains distinct demographic parameters). Understanding the spatial variation of populations, how they interact, and how demographic parameters differ among these populations are essential to sound management of focal species.

For example, sedentary stocks such as benthic invertebrates are typically structured as metapopulations; the subpopulations stay connected through larval or juvenile dispersal. The strong spatial effects not only make it difficult for a population to persist on its own, but adding in pressure from fishing has the chance to lead to stock depletion [67]. In Bristol Bay, sockeye salmon populations exist as mixed stocks (i.e., a metapopulation stock complex) during their adult phase. Management of salmon has historically focused on the metapopulation stock complex, rather than concentrating on the most productive populations. As a result, sockeye salmon harvest has remained relatively stable over decades. In the conservation of threatened species it is important to recognize that single populations have a high risk of extinction, and effectively managing for species persistence requires a metapopulation-level approach. For example, recovery strategies for Puget Sound Chinook salmon recommend two to four viable subpopulations within each geographic region to reduce the risk of extinction for the metapopulation [68].

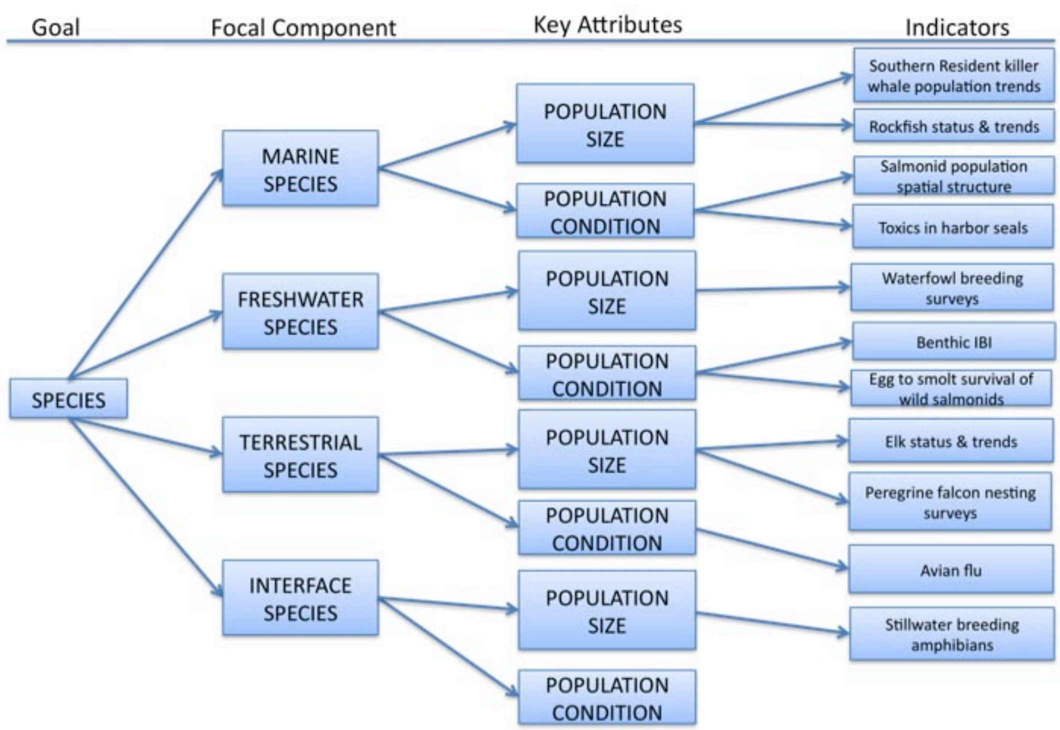


Figure 3. Summary of framework organization for Species goal. The list of indicators is illustrative only, and not complete.

Key Attributes – Food Webs

The food web indicator evaluations focused on two key attributes: (1) community composition, and (2) energetics and material flows. These two attributes reflect the structure and function of a food web and were drawn from a large literature on the subject [42, 69-74]. Food web attributes provide a measure of the extent to which different components of the ecosystem interact (e.g., habitats and species) along with important contextual information for understanding the status of the individual components themselves.

We have adopted a broad definition of community composition that includes species diversity, trophic diversity, functional redundancy, and response diversity. This definition is consistent with "community attributes," a key attribute for food webs recently designated by the PSP [37]. Species diversity encompasses species richness, or the number of species, in the food web, and species evenness, or how individuals or biomass are distributed among species within the food web [69]. Trophic diversity refers to the relative abundance or biomass of different primary producers and consumers within a food web [42]. Consumers include herbivores, carnivores or predators, omnivores, and scavengers. Functional redundancy refers to the number of species characterized by traits that contribute to a specific ecosystem function, whereas response diversity describes how functionally similar species respond differently to disturbance [75]. For example, a food web containing several species of herbivores would be considered to have high functional redundancy with respect to the ecosystem function of grazing, but only if those herbivorous species responded differently to the same perturbation (e.g., trawling) would the food web be considered to have high response diversity.

Like community composition, the second key attribute of food webs, energy and material flows, was previously highlighted by the PSP [37]. This attribute includes ecological processes such as primary production and nutrient cycling, in addition to flows of organic and inorganic matter throughout a food web. Primary productivity is the capture and conversion of energy from sunlight into organic matter by autotrophs, and provides the fuel fundamental to all other trophic transfer in a food web. Material flows, or the cycling of organic matter and inorganic nutrients (e.g., nitrogen, phosphorus), describe the efficiency with which a food web maintains its structure and function.

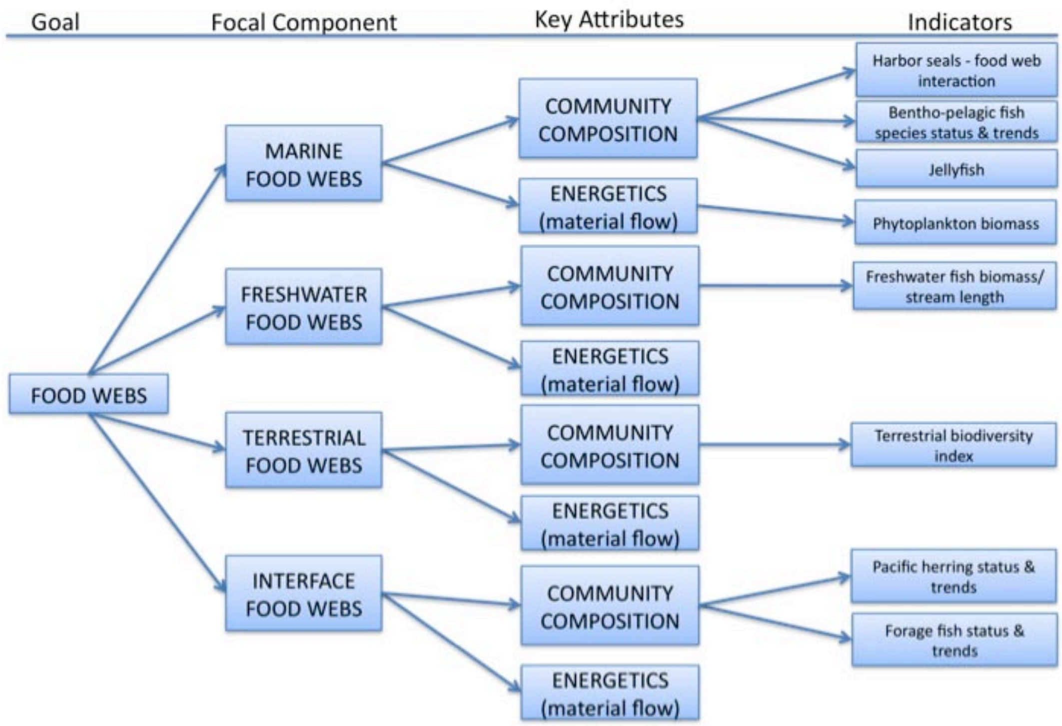


Figure 4. Summary of framework organization for Food Webs goal. The list of indicators is illustrative only, and not complete.

Key Attributes – Habitats

The Puget Sound basin encompasses diverse marine, nearshore, freshwater, and terrestrial habitats. As such, a key goal of the PSP is to have 'a healthy Puget Sound where freshwater, estuary, nearshore, marine, and upland habitats are protected, restored, and sustained' (from RCW 90.71.300). Many different ecological attributes may be used to describe habitat status and determine whether or not it is 'healthy'. The U.S. EPA (2002) identified various attributes of habitats (referred to as 'landscapes') including extent, composition, and pattern/structure; other attributes of habitats included dynamic structural characteristics and physical structure [42]. The U.S. EPA also acknowledged habitat condition, but recommended its use as a species attribute (i.e., habitat suitability) because they defined condition in terms of the organisms of interest [42]. Similar landscape attributes identified by the Heinz Center (2008) included extent and pattern [44].

In 2009, the PSP structured their reporting on ecosystem status around two broad indicator categories for the habitat goal: extent and condition of ecological systems [37]. These broad categories were selected to represent key attributes associated with the habitat goal [37], and were used to report on extent and condition of focal habitats in Puget Sound [76]. Simultaneously, a PSP working group identified several key habitat attributes including: estuarine wetlands, delta or river mouth condition, coastal embayments and lagoons, forage fish spawning habitat/substrate, condition of shorelines and condition of beaches, benthic condition, marine water condition, freshwater condition, spatial extent of ecological systems (terrestrial), condition of ecological systems or plant associations (terrestrial), and functional condition for key terrestrial species [37].

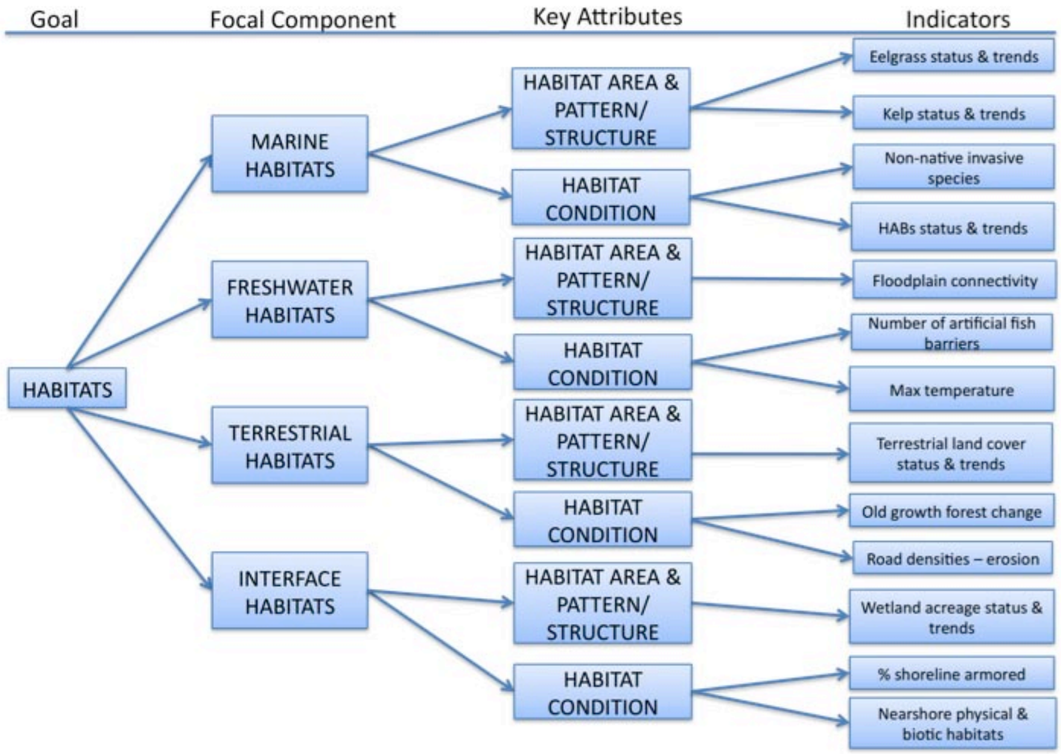


Figure 5. Summary of framework organization for Habitats goal. The list of indicators is illustrative only, and not complete.

Habitat area and pattern/structure combines several measures. Habitat area is defined as the areal extent and shape of each habitat type. Pattern/structure refers to the number of habitat types, the number of patches of each habitat, fractal dimension (i.e., habitat complexity), and connectivity. Habitat condition refers to abiotic properties (i.e., physical and chemical properties) and biotic properties (e.g., invasive or nuisance species, dominant species). Dynamic structural characteristics (i.e., changes in physical habitat complexity and morphology) are also included in habitat condition because they maintain the diversity of natural habitats. Water quality and benthic condition also contribute to habitat condition; however, according to the PSSU framework, they fall under the Water Quality goal and will therefore be discussed in that section.

Key Attributes – Water Quality

The purpose of the framework development with regard to indicator selection, was to ensure that there was complete coverage of the goals by the indicators. The first division of goals was into ecologically unique domains (e.g., marine water, freshwater, and ecotones), which defined the Key Attributes. The properties of the Key Attributes must be known in order to define the state of that aspect of the ecosystem. Key attributes must be managed in order to sustain each conservation target (i.e. focal components) [77, 78]. This approach is similar to that previously utilized by the PSP [37].

There are three key attributes, which articulate Water Quality: hydrodynamics, the physical and chemical parameters, and trace inorganic and organic contaminants. These key attributes for water quality have also been utilized elsewhere [42, 43, 79].

Hydrodynamics are important characteristics of water quality in marine, freshwater, and transitional (e.g., wetlands, estuaries, etc.) systems. River and stream hydrodynamics are defined by various aspects of the flow regime including magnitude, frequency, duration, timing, and rate of change. Each of these has important impacts on ecology and human health and well-being [80-83]. The hydrodynamics of river and stream is discussed in the Water Quantity section of this Puget Sound Science Update. Lake hydrodynamics are generally defined by mixing, stratification (i.e. the lack of mixing), and residence times. All of these are key aspects of nutrient cycling and can be deterministic in lake water quality [84, 85]. Hydrodynamics are also important in marine environments. Offshore circulation patterns and seawater intrusions into Puget Sound bring in nutrient rich waters, which can impact eutrophication and dissolved oxygen (see Chapter 2 of the Puget Sound Science Update; [86-90]). Rivers and streams entering Puget Sound create areas of density stratification, which can also affect eutrophication [90, 91]. Hydrodynamics are critical in understanding water quality and have been incorporated as a Key Attribute.

Physical and chemical parameters are also crucial in determining water quality. The suitability of freshwater and marine water systems to support biota is strongly dependent on temperature and dissolved oxygen (DO; see [92, 93] and references therein). Low DO is an issue of management importance in the Hood Canal and the south Puget Sound [94]. The level of nutrients such as nitrogen and phosphorus in lakes and estuaries can affect primary productivity and habitat quality [86, 95-101]. Anthropogenic nutrient inputs have been associated with harmful algal blooms (see Chapter 2 of the Puget Sound Science Update; [102]). Increasing levels of atmospheric carbon dioxide in the may lead to decreased pH with ocean acidification, potentially resulting in severe impacts on key marine organisms with calcium carbonate exoskeletons [103]. General physical and chemical parameters are of import in defining water quality and are, thus, utilized as Key Attributes.

The presence and concentrations of trace organic and inorganic chemicals, also known as toxics, contaminants, pollutants, etc., may have impacts of the human health and the environment. Much of the implementation of the Clean Water Act has focused on the reduction of chemicals into surface waters for "the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water" [104]. A discussion of the toxic contaminants in Puget Sound is included in Chapter 2 of this Puget Sound Science Update, and also Section 5.4. Due to their potential importance both ecologically and to human-well being, trace organic and inorganic chemicals is a Key Attribute of water quality.

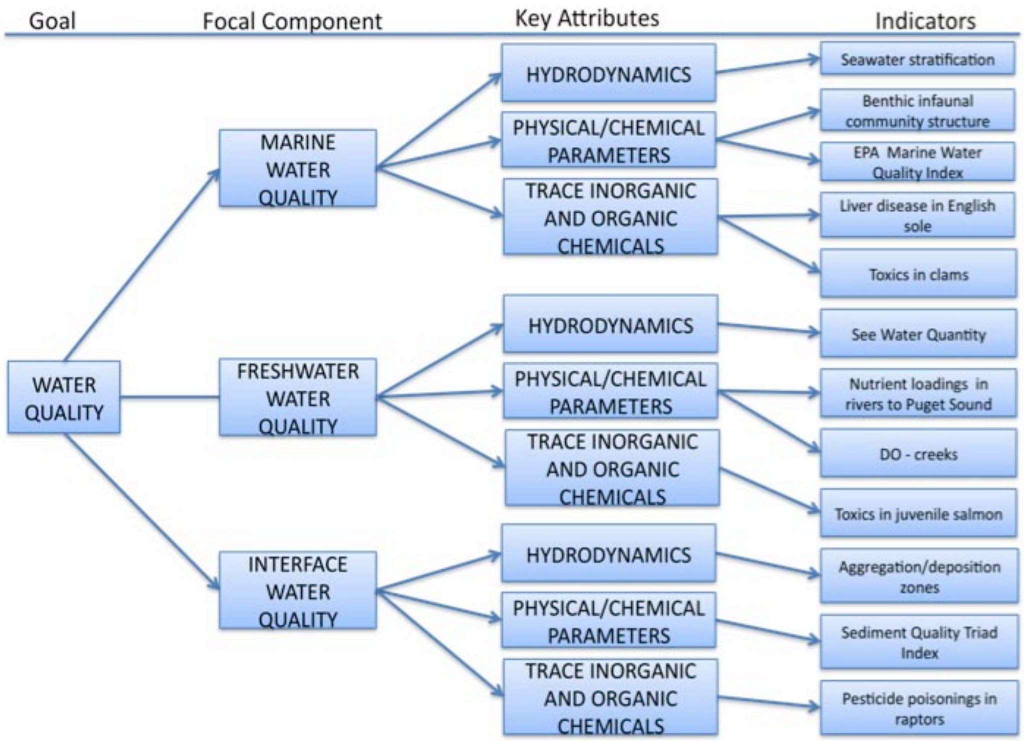


Figure 6. Summary of framework organization for Water Quality goal. The list of indicators is illustrative only, and not complete.

Key Attributes – Water Quantity

In order to evaluate indicators of water quantity, we used three distinct Key Attributes: the surface water hydrologic regime, groundwater levels and flows, and consumptive water use and supply. The PSP has utilized other organizational frameworks though they selected similar attributes. In the 2009 document, "Identification of Ecosystem Components and Their Indicators and Targets," water quantity was not dealt with as an explicit goal but rather as supportive of habitats and human uses [37]. This resulted in the selection of freshwater extent, freshwater condition, and water supply for end users as attributes – all similar to the Key Attributes used herein. The EPA defined surface and groundwater flows as an essential ecosystem attribute category with subcategories including pattern of surface flows, hydrodynamics, and pattern of groundwater flows [42]. Their framework focused on ecological condition and did not explicitly include human dimensions. The Heinz Center reports on the extent of freshwater ecosystems, changing stream flows, water withdrawals, and groundwater levels [44]. Other studies have reported the use of similar attributes to define the state of water quantity [105].

The surface water hydrologic regime has important impacts on the regional ecosystems (see [80] and references, therein). The groundwater is an important source both for consumptive use and river and stream base-flows. Consumptive water use and supply are important measures of resource conservation and supply and relate strongly to the human health and well-being of the region.

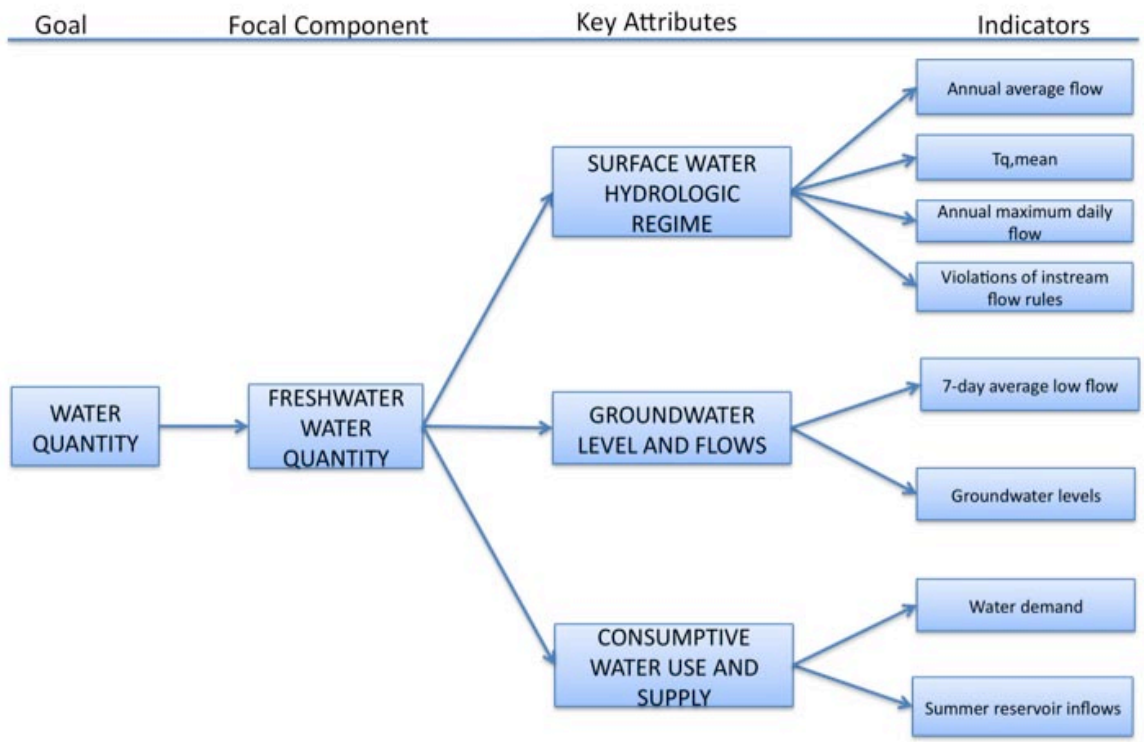


Figure 7. Summary of framework organization for Water Quantity goal. The list of indicators is illustrative only, and not complete.

Tier 4: Indicators.

Indicators are metrics that reflect the structure, composition, or functioning of an ecological system [42, 44]. Indicators are measurable characteristics that can assess changes in ecosystem attributes. A list of candidate indicators was selected from several sources (see Section 4.1) and each indicator was assigned to a specific Key Attribute based on expert opinion. Indicator identification and evaluation is discussed in Section 4.

A conceptual framework for selecting indicators of ecosystem condition is valuable for several reasons. First, indicators are often selected based on the degree to which they meet a number of criteria individually, rather than on the basis of how they collectively assess ecosystem condition [33]. A conceptual framework explicitly includes the inter-relation of indicators as part of the indicator selection process, and helps to develop consistent indicator sets [33]. Second, a conceptual framework provides flexibility. For example, if the goal is to assess marine ecosystem health using only ten indicators, a hierarchical framework provides a way to select indicators so that all the relevant ecosystem components are included. In this case, one to three indicators would be selected from Marine Species, Marine Food Webs, Marine Habitats, and Marine Water Quality in order to ensure adequate representation of all the important features. Third, a framework highlights indicators that may be relevant to multiple goals, focal components, or attributes. For example, the population abundance of Western sandpipers is related to the Species goal, but may also be relevant to the Habitats goal if their abundance reflects changes in habitat condition. Finally, a framework explicitly links indicators→attributes→ focal components→goals, which ensures sufficient coverage of the Key Attributes essential to each goal. A conceptual framework provides a structured yet flexible way to select indicators that best represent the environmental issue at hand.

Key point: A carefully crafted framework provides a robust means for assuring that ecosystem indicators are explicitly linked to societal goals. The approach we present melds a number of separate PSP activities into a single, transparent framework and provides a structured yet flexible means to select ecologically and socially meaningful indicators.

Evaluation of Potential Indicators for Puget Sound

1. Indicator selection and organization

We began our evaluation of indicators by compiling a list of available indicators. To build on previous efforts, we selected indicators from three sources: a 2008 report titled, "Environmental Indicators for the Puget Sound Partnership: A Regional Effort to Select Provisional Indicators (Phase 1);" the PSP Action Agenda; and the 2009 PSP Technical Memoranda, "Identification of Ecosystem Components and Their Indicators and Targets," and "Ecosystem Status and Trends" [1, 34, 37, 76]. Further, a small number of indicators were identified through a review of the regional literature (e.g., [23, 106]) and were also included on the list of available indicators.

The authors of the "Environmental Indicators for the Puget Sound Partnership" report reviewed over 100 documents to create a list of more than 650 indicators that had been proposed or used in Puget Sound and Georgia Basin [34]. Using a set of screening criteria, they reduced the list to approximately 250 indicators that were "good," or "potential." Further, there was a set of indicators, which were of, "possible future," value but were not considered for use in that evaluation because they did not have existing data. However, they were included in our evaluation. Finally, there was a small group of indicators indentified that were not evaluated in the 2008 work. These were also included in the PSSU process.

The PSP Action Agenda listed a subset of environmental indicators, which had been selected based on a review by the PSP Science Panel [1]. This list of 102 indicators was included in our evaluation process to ensure completeness.

In 2009, the PSP began a separate indicator selection process specifically guided by the Open Standards for the Practice of Conservation [3, 37] which included the development of Focal Components and Key Attributes through a series of workshops. As summarized in the 2009 Technical Memorandum, "Identification of Ecosystem Components and Their Indicators and Targets," the process resulted in the identification of over 160 indicators, including many associated with the Built Environment, Working Marine Industries, Working Resource Lands and Industries, Nature Oriented Recreation, and Aesthetics, Scenic Resources, and Existence Values [37]. These indicators were included in our evaluation, unless they had been previously evaluated and found to be theoretically unsound [34].

In a parallel effort, the PSP Technical Memorandum, "Ecosystem Status and Trends," reported on a set of 43 indicators [107]. A subset of these were used in the 2009 State of the Sound report. All were included for consideration.

Finally, with specific regard to the indicators of Water Quantity, the literature identifies well over 150 unique indicators, which can be utilized to track various aspects of the hydrologic flow regime (see [108]). Instead of individually evaluating each indicator, a literature review was undertaken to identify issues of potential concern in the Puget Sound region (see Section 5.5) and the results of that literature review were used to focus the choice of Water Quantity indicators for further evaluation.

The entire set of indicators was combined and redundant indicators removed, yielding a composite list of over 250 preliminary indicators for evaluation. The indicators were then organized according to the Key Attributes of our framework (see Figure 2 in Section 3). Our initial organization was based solely on expert opinion and recommendations. The process identified several indicators that could be appropriately categorized under more than one Key Attribute. However, the evaluation process allowed for the reorganization or reassignment of indicators based on the results of the review of the literature.

Once organized, each individual indicator was evaluated against a set of evaluation criteria, as described below. Importantly, the aim of this process was to support the science-policy processes of the PSP by evaluating the degree to which indicators meet

Indicator Evaluation Criteria

There exist nearly as many guidelines and criteria for developing and selecting individual indicators as there exist indicators. The summary of criteria for relevant and reliable indicators builds on the recommendations in the indicator report to the PSP [34], and is based on [29, 30, 33, 41, 43, 109-115]. These criteria apply to indicators of ecosystem state, the focus of this chapter. However, the approach and criteria we develop here is immediately transferable to the rigorous evaluation of driver and pressure indicators as well.

We divide indicator criteria into three categories: primary considerations, data considerations, and other considerations. Primary considerations are essential criteria that should be fulfilled by an indicator in order for it to provide scientifically useful information about the status of the ecosystem in relation to PSP goals. Data considerations relate to the actual measurement of the indicator. Data considerations criteria are listed separately to highlight ecosystem indicators that meet all or most of the primary considerations, but for which data are currently unavailable. Other considerations criteria may be important but not essential for indicator performance.

Other considerations are meant to incorporate non-scientific information into the indicator evaluation process. Ecosystem indicators should do more than simply document the decline or recovery of ecosystem health, they must also provide information that is meaningful to resource managers and policy makers [8]. Because indicators serve as the primary vehicle for communicating ecosystem status to stakeholders, resource managers, and policymakers, they may be critical to the policy success of EBM efforts, where policy success can be measured by the relevance of laws, regulations, and governance institutions to ecosystem goals. Importantly, policy success does not necessarily produce effective management since it is possible to be successful at implementing poor policy. Nonetheless, advances in public policy and improvements in management outcomes are most likely if indicators carry significant ecological information and resonate with the public.

It should be noted that all of the criteria listed need not be weighted equally, nor is it necessary to meet all of the criteria for an indicator to be valuable or of use for a specific application. Scientifically credible indicators should meet the "primary considerations" we outline below, and that further selection and evaluation be based on local needs and guided by the data and other considerations. A discussion of potential ranking is in Section 5.6.

The criteria we used are as follows:

Primary considerations

1. **Theoretically-sound:** Scientific, peer-reviewed findings should demonstrate that indicators can act as reliable surrogates for ecosystem attribute(s)
2. **Relevant to management concerns:** Indicators should provide information related to specific management goals and strategies.
3. **Responds predictably and is sufficiently sensitive to changes in a specific ecosystem attribute(s):** Indicators should respond unambiguously to variation in the ecosystem attribute(s) they are intended to measure, in a theoretically- or empirically-expected direction.
4. **Responds predictably and is sufficiently sensitive to changes in a specific management action(s) or pressure(s):** Management actions or other human-induced pressures should cause detectable changes in the indicators, in a theoretically- or empirically-expected direction, and it should be possible to distinguish the effects of other factors on the response.
5. **Linkable to scientifically-defined reference points and progress targets:** It should be possible to link indicator values to quantitative or qualitative reference points and target reference points, which imply positive progress toward ecosystem goals.
6. **Complements existing indicators:** This criterion is applicable in the selection of a suite of indicators, performed after the evaluation of individual indicators in a post-hoc analysis. Sets of indicators should be selected to avoid redundancy and increase the complementary of the information provided, and to ensure coverage of Key Attributes.

Data considerations

1. **Concrete:** Indicators should be directly measureable.
2. **Historical data or information available:** Indicators should be supported by existing data to facilitate current status evaluation (relative to historic levels) and interpretation of future trends.
3. **Operationally simple:** The methods for sampling, measuring, processing, and analyzing the indicator data should be technically feasible.
4. **Numerical:** Quantitative measurements are preferred over qualitative, categorical measurements, which in turn are preferred over expert opinions and professional judgments.
5. **Broad spatial coverage:** Ideally, data for each indicator should be available in all PSP Action Areas.
6. **Continuous time series:** Indicators should have been sampled on multiple occasions, preferably without substantial time-gaps between sampling.
7. **Spatial and temporal variation understood:** Diel, seasonal, annual, and decadal variability in the indicators should ideally be understood, as should spatial heterogeneity/patchiness in indicator values.
8. **High signal-to-noise ratio:** It should be possible to estimate measurement and process uncertainty associated with each indicator, and to ensure that variability in indicator values does not prevent detection of significant changes.

Other considerations

1. **Understood by the public and policymakers:** Indicators should be simple to interpret, easy to communicate, and public understanding should be consistent with technical definitions.
2. **History of reporting:** Indicators already perceived by the public and policymakers as reliable and meaningful should be preferred over novel indicators.
3. **Cost-effective:** Sampling, measuring, processing, and analyzing the indicator data should make effective use of limited financial resources.
4. **Anticipatory or leading indicator:** A subset of indicators should signal changes in ecosystem attributes before they occur, and ideally with sufficient lead-time to allow for a management response.
5. **Regionally/nationally/internationally compatible:** Indicators should be comparable to those used in other geographic locations, in order to contextualize ecosystem status and changes in status.

Indicator Evaluation Process

After constructing the framework, the explicit definition of the evaluation criteria, and the selection and organization of the individual indicators, each indicator was evaluated individually. Our intent was to assess each indicator against each evaluation criterion by reviewing peer-reviewed publications and reports. We chose this benchmark because it is consistent with the criterion of peer-review used by other chapters of the Puget Sound Science Update, and it is a criterion that is relatively easy to apply in a consistent fashion. However, we do recognize the value of non-peer reviewed documents as well as the opinion of expert panels. Consequently, where we found such documentation, we include it, while noting that it is not peer-reviewed. The result is a matrix of indicators and criteria that contains specific references and notes in each cell, which summarize the literature support for each indicator against the criteria. We reiterate here that our goal is to review and evaluate indicators that could inform the policy-science process underway in the Puget Sound Partnership. We do not recommend a final indicator portfolio.

Some specific points on the evaluation process:

1. The intent of including references was to provide sufficient evidence that the indicator met (or failed to meet) each of the specific evaluation criteria. Based on the references, an independent evaluator should be able to understand the important points of the process.
2. As is the standard for the entire PSSU, we required references to be peer-reviewed publications or reports. Internal agency documents were included when it was clear that there had been an explicit peer-review process.
3. There was a preference for literature based on studies conducted in the Puget Sound region.
4. The evaluation notes were meant to be of sufficient detail to allow an independent evaluator to understand the basis for conclusion, when it was not otherwise obvious from the references.
5. Each of the indicators was evaluated against a specific Key Attribute, which they were meant to describe. If, however, the detailed evaluation indicated that the indicator better

described a different Key Attribute, then the individual reviewing that indicator was given the liberty to reassign the indicator.
6. In some instances no references were found relating an indicator to a specific criterion. These cells were left blank.
7. Some of the Data Considerations were evaluated by a simple yes/no response when the conclusion was obvious (e.g., concrete, historical data, operationally simple, numerical, spatial coverage, continuous).

Certain criteria proved to be problematic during the evaluation. These included:

1. **Relevant to Management Concerns.** It was not always obvious to a reviewer if a particular indicator was relevant to management concerns. Management concerns were not always clearly documented or lacked specificity. Often, PSP background documents were referenced based on the presumption that they accurately reflected management concerns.
2. **Understood by Public and Policy Makers.** There is a lack of literature documenting the degree to which citizens or their representatives understand the meaning or intent of specific ecosystem indicators (or ecological concepts). The evaluation of an indicator under this criterion is often presumptive and may vary depending on the reviewer.
3. **Cost Effective.** The value of the information from an indicator was difficult to determine. Cost effectiveness may be measured by the value of decisions made based on the new information from the indicator. This is difficult because not only are decision scenarios complex and difficult to evaluate on a cost basis, but it is also difficult to predict the range of potential decisions that could be made based on the new information. Further, cost effectiveness may be measured by the opportunity cost of choosing one indicator over another. Assuming that the suite of indicators (and information) is limited, the value of choosing one indicator over another is not only related to the new information gained, but also the cost of the information lost by not collecting data for other indicators.
4. **Complements Existing Indicators.** It was necessary to have a complete suite of indicators in order to evaluate the complementarity and/or redundancy of each of the indicators. As mentioned above, this criterion should be applied in a post-hoc analysis.

Key point: Indicators should be evaluated using widely accepted and transparent criteria. This chapter used criteria derived from the vast literature on ecosystem indicators, which were divided into three groups: 1) Primary considerations are essential criteria that should be fulfilled by an indicator; 2) Data considerations relate to the actual measurement of the indicator; 3) Other considerations criteria may be important but not essential for indicator performance.

Next Step: Evaluations were focused on the presence or absence of peer-reviewed evidence that an indicator met each criterion. Thus, we did not evaluate the rigor of the evidence. An important next step will be to carefully review the evidence and distinguish between weak and strong evidence.

Results of the Indicator Evaluations

Detailed spreadsheets showing the results of the indicator evaluation are available at the following link: Indicator Spreadsheets. Summary tables are included at the end of this section. Following the framework outlined in Section 3, we organize the results of the evaluation by PSP ecosystem goals (i.e. Species, Habitat, Food Webs, Water Quality, and Water Quantity). Each goal has been divided per unique ecosystem domain (marine, freshwater, interface, and terrestrial).

A focused discussion of the evaluation by goal is presented in the following sections. The discussions include a summary of the results of the evaluation, as well as a presentation of the salient issues to Puget Sound, which were identified during the literature review. The section on Water Quality includes the complete literature review, which was performed in order to identify indicators appropriate for use in Puget Sound.

1. Species Evaluation

This version of the Puget Sound Science Update provides an initial evaluation of species indicators, but is not intended to be comprehensive. Focal species identified by O'Neill et al. [34] were evaluated as either measures of population size or population condition. Many of these were identified as potentially good species indicators, and several may be relevant to key attributes of the other PSP goals (e.g., habitat condition).

- The inclusion of more candidate freshwater and interface indicators, as well as indicators for population condition of marine and terrestrial species
- Evaluation of population condition indicators other than those related to organism condition (e.g., age structure, population structure)
- Explicitly defining vague indicators (e.g., insect species)||

Commonly used data sources to evaluate species indicators included: Washington Departments of Ecology, Fish and Wildlife, and Natural Resources, NMFS, USFWS and USGS.

Indicators of population size

We focused on three metrics of population size: the number of individuals in a population, total biomass, and population dynamics. Population abundance and biomass data are key measures of the overall health of a focal species. Insight into the status and trends of a focal species can also be used to infer changes in ecosystem structure and function. While population size can be used to assess population viability, more accurate predictions of viability can be obtained by including the mechanisms responsible for the dynamics of the population. Population dynamics thus provide a predictive framework to evaluate the combined effect of multiple mechanisms of population regulation (e.g., birth and death rates, immigration and emigration) to evaluate changes in abundance through time. The Washington Departments of Ecology, Fish and Wildlife, and Natural Resources, USGS, and NMFS, among others, have ongoing monitoring efforts of population status and trends for numerous species throughout the sound.

The use of species attributes by the PSP has largely been limited to population size. For example, in the 2007 and 2009 State of the Sound documents only measures of population size were reported for all species indicators (except salmon) [40, 116]. While the PSP has historically recognized the importance of monitoring species health or condition, their use of 'condition' was limited to measurements of toxic contaminants in various species, and was meant to be an indicator of Water Quality (see [40, 116]). In the following section we discuss the utility of population condition as an independent attribute for assessing the status of focal species in Puget Sound.

Indicators of population condition

Whereas the preceding attribute is concerned with measures of population size, there are instances when the "health" of the population may be of interest. For example, monitoring changes in population condition may presage an effect on population size or provide insight into long-term population viability. The dynamics of many populations are better understood through knowledge of population condition such as organism condition, age structure, genetic diversity, phenotypic diversity, and population structure. Impaired condition of any or all of these subcategories indicates biological resources at risk.

Organism condition represents both physiological and disease status. Physiological status reflects the general condition of an organism whereas disease status signals the presence of harmful agents. Thus monitoring changes in organism condition can be used to infer changes in environmental conditions. Population age structure is used to evaluate long-term stability and viability of a population by modeling trends through time. Genetic diversity measures are important in assessing population condition because loss of genetic variation can reduce individual fitness as well as the ability of populations to evolve in the future [62]. Phenotypically diverse populations (i.e., each subset of the total population is adapted to a slightly different environmental condition) have an increased capacity for adapting to changing environmental conditions, which can be vital for long-term population sustainability. Similarly, insight into population structure (i.e., how different populations interact in space) can be useful for predicting the effects of changing conditions on population viability. WDFW and NMFS monitoring programs (among others) provide important information for assessing population conditions.

Evaluation of species indicators in Puget Sound

There were seventy-seven species indicators identified by O'Neill et al. [34]and of these, we have evaluated sixty. The majority of those evaluated are indicators of population size for marine and terrestrial species. Several focal components would benefit from indicator development including Interface Species (population size and condition), Freshwater Species (population size and condition), and Terrestrial Species (population condition only). The current status of indicator evaluations for each species focal component is summarized below.

Marine species indicator evaluation

Population size. There were twenty-nine indicators of marine species population size identified (Table 4). Most of these indicators are conceptually valid, and about half those evaluated were an overall good indicator of species abundance. There were several good indicators relevant to food webs as well as key attributes for other PSP goals (e.g., habitat condition). Valuable data sources for assessing marine species abundance included (among others) WDFW, WDOE, WDNR, USGS and USFWS, and NMFS.

In general, indicators that did not perform well failed because:

- Data are unpublished, poorly documented or does not exist
- Unable to assess whether they respond predictably to ecosystem attributes or to management actions or pressures
- Variation is not well understood, especially for migratory species

Indicators that performed well against all criteria included: total run size of salmonids (hatchery and wild), salmon and steelhead status and trends, marine bird aerial estimates (non-breeding populations), and pinto abalone status and trends. Pinto abalone is a unique indicator because, while it performs well against most criteria, is not necessarily theoretically-sound. A study by Rothaus et al. (2008) concluded that declines in abalone abundance are not likely to recover due to historic overharvesting, making it a poor indicator for healthy and sustaining species [117].

Table 4. Summary of Marine Species - Population Size indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Pinto abalone status & trends has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Mammals	Southern Resident killer whale population trends	3	4	3	Overall good indicator of species (e.g., vital sign) but may not be best indicator of ecosystem structure & function. Also, does not respond predictably to management actions.
	Gray whale status & trends	3	3	0	May serve as a good indicator of species abundance. Difficult to determine impacts of management actions b/c long-lived and migratory. Gray whales may be an indicator for climate change if migratory patterns shift.
	Harbor porpoise/Dall's porpoise status & trends	0	2	1	No evidence to recommend this as an indicator. Few studies have looked at abundances in Puget Sound, they are highly migratory, and very little information is known about populations.
	Harbor seal status & trends	3	8	3	Overall good indicator of species abundance (e.g., vital sign), but not necessarily food webs. To avoid redundancy, choose between this indicator and harbor seal – food web interaction.
Key Fish	Total run size of salmonids (hatchery & wild)	5	8	4	Overall good indicator; peer-reviewed literature supporting most criteria.
	Harvest of wild salmonid populations	Not yet evaluated			
	Exploitation rates of wild salmonid populations	Not yet evaluated			
	Marine bottomfish harvest	2	3	3	Theoretically-sound however, unable to determine response to ecosystem attributes or management actions/pressures. MSY estimates lacking for many marine bottomfish in PS.
	Rockfish status & trends	1	3	1	Rockfish notes as best indicator for some ecosystem attributes [118], but due to life history characteristics, it is difficult to assess whether they respond predictably to ecosystem attributes or management. Historical harvest data available.
	Salmon & steelhead status & trends	5	8	4	Overall good indicator; peer-reviewed literature supporting most criteria.
	Marine resident fish species status & trends	0	0	0	Information does not exist for several of the species suggested. Rationale for collecting these data needs to be further evaluated prior to developing this indicator.
Birds	Marine waterfowl harvest	0	0	1	Theoretically-unsound. Mostly unpublished data; marine waterfowl population numbers are not well documented, so difficult to determine the effects of harvest on overall abundances.
	Marine bird aerial estimates – non-breeding populations	3	8	4	Overall good indicator of species abundance (no relevance to food webs). Long history of reporting that covers virtually all PSP action areas. Because a mix of residents and migrants, changes in abundance could be the result of pressures outside PS.
	PIGU nesting colony trends	0	0	0	Poor indicator. Difficult to find any peer-reviewed literature on pigeon guillemot population numbers or nesting colony trends.
	Marine bird breeding abundance	1	0	0	Poor indicator. There are Canadian seabird breeding datasets; equivalent datasets lacking for Puget Sound.
	Black oystercatcher abundance	4	3	2	Good theoretical species indicator however, patchy surveys of varying levels of sampling effort, coverage, and methodologies preclude formal comparison of data. Also, not present in southern and central Puget Sound.
	Marine bird fishing mortality	3	2	0	Theoretically-sound and relevant, but scattered reporting of bycatch in local fisheries. Complicated data analysis.
	Glaucous wing gull abundance at nesting colonies	1	4	2	Theoretically-sound but does not meet any other Primary Considerations. Data available, mostly for north Puget Sound. Not particularly cost-effective and in general, not locally appreciated.
	Marine birds – shore-based estimates of non-breeding populations	Not yet evaluated			
	Western sandpiper status & trends	2	3	2	Good species indicator and may also be a good indicator of habitat condition. Habitat loss is identified as main pressure, but difficult to ascertain what the impact has been on population abundance. Trend analysis of data is absent.
	Scoter & Harlequin ducks – non-breeding populations	3	3	1	May be a good indicator of species and food webs b/c they follow herring spawning. Unpublished data sets that are regionally patchy; variation in local trends not well understood.
	Cormorant abundance at nesting colonies	2	6	2	May be a good species and food web indicator; Slater & Byrd (2009) found bird abundance to predict changes in marine food webs [119]. Long-term monitoring programs so good data availability.
Shellfish & Other Inverts	Dungeness crab abundance	1	2	2	Theoretically-sound but does not meet any other Primary Considerations. Abundance is measurable through pre- and post-season crab pot surveys but no published data available.
	Dungeness crab harvest	2	6	4	May be a good indicator b/c theoretically-sound and relevant to management, but year-to-year variation in harvest is not well-understood. Long-term data available from harvest report cards.
	Pinto abalone status & trends	3	6	4	Long-term data available and relevant to management, but Rothaus et al. (2008) concludes that declines in abundance are not likely to recover due to historic overharvesting [117].
Plants	Eelgrass status & trends	Evaluated under Marine Habitats			
	Kelp status & trends	Evaluated under Marine Habitats			
	Marine macro algae status & trends	Evaluated under Marine Habitats			

Population condition. There were fifteen indicators of population condition (Table 5). Most indicators were based on measures of organism condition, with considerably fewer indicators representing the other measures of population condition (i.e., age structure, population structure, phenotypic diversity, and genetic diversity). In the future, candidate indicators may need to be developed for these additional measures of population condition, especially as they relate to focal species of management concern.'''

Many of the indicators of organism condition (e.g., toxics in mussels) listed were evaluated under Marine Water Quality; we decided that for the purposes of this document, contaminant-related indicators in lower trophic level organisms provided pertinent information on water condition. Future iterations of the PSSU may choose to evaluate such indicators in relation to species condition, especially as the science develops to support the idea of population-level effects [120]. The remaining four indicators evaluated under marine species population condition were theoretically-sound, and all but one (marine bird mortality) performed well against all criteria. These included: toxics in harbor seals, liver disease in English sole, and toxics in adult Chinook and coho salmon. Data sources mainly used to evaluate organism condition included WDFW, NMFS, Cascadia Research, Fisheries and Oceans Canada, and past PSAT reports. Several indicators including smolt to adult return for wild salmonids, salmonid diversity, star protein/DNA damage in fish, abnormal fish embryonic development, marine growth and survival of juvenile coho, and salmonid population spatial structure still need to be evaluated.

Table 5. Summary of Marine Species - Population Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Marine bird mortality has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Mammals	Toxics in harbor seals	4	7	3	Good indicator but more sites are needed for Puget Sound.
Key Fish	Smolt to adult return for wild salmonids	Not yet evaluated			
	Salmonid diversity	Not yet evaluated			
	Liver disease in English sole – see also Marine Water Quality	5	8	5	Populations with elevated liver disease show symptoms of reproductive impairment and age-selected mortality. Changes in prevalence of liver disease are used to document improvements in fish health. Thresholds for PAH levels in sediment associated with increased prevalence have been defined. There is historic coverage of over 50 sites, currently limited to 8 with representing urban, near-urban and non-urban site.
	Vtg induction in male fish	Evaluated under Marine Water Quality			
	Star protein/DNA damage	Not yet evaluated			
	Abnormal embryonic development	Not yet evaluated			
	Toxics in adult Chinook & coho salmon – see also Marine Water Quality	3	7	4	May be a good overall indicator of species condition (e.g., vital sign indicator), but does not respond predictably to management actions or pressures. Long-term monitoring program for Chinook salmon was discontinued in 2006. Risk to fish health will go down with lower contaminant levels.
	Toxics in adult Pacific herring – see also Marine Water Quality				Impairment to fish health increase with toxic levels in fish. Thresholds for toxics have been defined adult herring. Sampling requires technical expertise and equipment. Historic coverage major herring populations. Continuous time series for three populations from 1999.
	Marine growth & survival of juvenile Coho	Not yet evaluated			
	Salmonid population spatial structure	Not yet evaluated			
	Toxics in English sole – see also Marine Water Quality				Elevated contaminant levels in English sole (including PAH metabolites in bile) increase with concentrations in the environment and elevated levels are associate with liver disease and reproductive impairment. Thresholds exist for some chemicals. Sampling requires specialized techniques and instrumentation. Historic coverage of over 50 sites, currently limited to 8 sites representing urban, near-urban and non-urban.
Birds	Marine bird mortality	2	8	2	Data has been collected all over Puget Sound since 2000. Theoretically-sound and responds to management efforts to reduce seabird bycatch. Underappreciated by management and the public b/c lower number of dead birds generally found in the sound.
Shellfish & Other Inverts	Benthic infaunal community structure (sediment quality)	Evaluated under Marine Water Quality			
	Toxics in mussels – see also Marine Water Quality				Threshold specific to the health of mussels are not know. Continuous coverage from mid 80's.

Freshwater species indicator evaluation

Population size. There were five indicators of freshwater species population size identified (Table 6). Of these, three have not been evaluated (mammal species, total number of spawning adult salmonids, and freshwater resident fish species). The remaining indicators, waterfowl status and trends of midwinter populations and waterfowl breeding surveys, both performed poorly. WDFW, USFWS, and the Pacific Flyway Council provide overviews of waterfowl population status and trends in the Pacific flyway region, however there are no specific references to Washington populations.

Also of note, mammal species and freshwater resident fish species may need to be more explicitly defined before they are evaluated.

Table 6. Summary of Freshwater Species - Population Size indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Waterfowl breeding surveys has peer-reviewed literature supporting 0 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Mammals	Mammal species	Not yet evaluated			
Key Fish	Total number of spawning adult salmonids (hatchery & wild)	Not yet evaluated			
	Freshwater resident fish species	Not yet evaluated			
Birds	Waterfowl status & trends of midwinter populations	0	0	0	Currently a poor indicator; references provided by WDFW and USFWS provide an overview of waterfowl population status & trends, but no specific references for WA midwinter populations. More specific information is needed.
	Waterfowl breeding surveys	0	0	0	Currently a poor indicator; references provided by WDFW and USFWS provide an overview of waterfowl population status & trends, but no specific references for WA breeding surveys. More specific information is needed.

Population condition. Six indicators of freshwater species population condition were identified (Table 7), and only one indicator (salmonid population growth rate) has currently been evaluated in this section; it received an overall good rating across all the criteria with references primarily from NMFS. Three indicators, toxics in juvenile salmon, benthic IBI and aquatic vertebrate IBI, are evaluated under Water Quality though they do pertain to population condition as well. Two remaining indicators, recruits per spawner of wild salmonids and egg to smolt survival of wild salmonids, need to be evaluated.

Table 7. Summary of Freshwater Species - Population Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Salmonid population growth rate has peer-reviewed literature supporting 5 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Key Fish	Recruits/spawner of wild salmonids	Not yet evaluated			
	Egg to smolt survival of wild salmonids	Not yet evaluated			
	Salmonid population growth rate	5	8	4	Overall good indicator; peer-reviewed literature supporting most criteria.
	Toxics in juvenile salmon – see also Interface Water Quality				Chinook salmon is an ESA listed species in Puget Sound. Age of the fish will determine whether local or regional water quality is reflected. Health-effects thresholds exist for PCBs and TBT. No consistent monitoring program in Puget Sound, however, multiple studies provide baseline data.
Shellfish & Other Inverts	Benthic IBI – macro-invert communities	Evaluated under Freshwater Quality			
Key Species	Aquatic vertebrate IBI	Evaluated under Freshwater Quality			

Terrestrial species indicator evaluation

Population size. There were nineteen indicators of terrestrial species population size identified (Table 8). Twelve of these indicators are conceptually valid, and about half may be good overall indicators of species abundance. Several indicators may provide relevant information to key attributes for other PSP goals (e.g., habitat area and condition). Data from WDFW and USGS Breeding Bird Survey (BBS) provided nearly all of the information on terrestrial species abundance. The residual indicators generally performed poorly because:

- Data coverage is limited
- Unable to determine relevance to management or response to management actions or pressures
- Tracking or monitoring species abundance is particularly difficult

Indicators that performed relatively well against all criteria included: terrestrial game species harvest, terrestrial breeding bird counts, terrestrial bird species, and Christmas bird counts. Several indicators including deer population status and trends, elk status and trends, backyard wildlife population trends, bald eagle status and trends, cavity nesting birds, Taylor's checkerspot butterfly, and marbled murrelets also performed relatively well against the primary considerations, but failed most of the data and other considerations criteria.

The majority of indicators that did well against the criteria are either mammals or birds, and it may be useful to develop candidate indicators for underrepresented or absent guilds (e.g., insects, plants).

Table 8. Summary of Terrestrial Species - Population Size indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Upland plant

species has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Mammals	Mountain goat status & trends	0	0	0	Poor indicator due to the inherent difficulties in tracking them.
	Deer population status & trends	3	3	4	Theoretically-sound and relevant to management. Population estimates are derived from harvest statistics (WDFW). Good species indicator and may also provide information on food webs and habitat condition since ungulate abundance can significantly affect ecosystem structure and function through browsing pressure.
	Elk status & trends	3	3	4	Theoretically-sound and relevant to management. Population estimates are derived from harvest statistics (WDFW). Good species indicator and may also provide information on food webs and habitat condition since ungulate abundance can significantly affect ecosystem structure and function through browsing pressure.
	Backyard wildlife population trends	3	1	1	May be a good species indicator, although evidence for management relevance is lacking (but may be used to encourage) citizen action). Monitoring data sources are likely to be widely dispersed and patchy in time.
	Terrestrial game species harvest	3	7	4	Overall good indicator; peer-reviewed references support many of the criteria.
	Mammal species	0	0	1	Currently a poor indicator b/c compilation of species where some are more extensively monitored than others. Possibly link this indicator with issues of landscape connectivity (of particular importance to mammals) to evaluate progress of landscape planning and assessment strategies.
Birds	Terrestrial breeding bird counts	3	6	4	Theoretically-sound and long-term data available. Difficult to determine management relevance or response to management actions/pressures. Phenological timing of migrations may serve as a useful leading indicator of climate change impacts.
	Peregrine falcon nesting surveys	3	3	4	Does not appear to be a good indicator (theoretically-unsound); lack of data in Puget Sound and variations in abundance not well understood.
	Bald eagle status & trends	5	3	2	Overall good species indicator (e.g., vital sign) although data coverage and variability not well documented in Puget Sound.
	Band-tailed pigeon mineral site counts	0	3	0	Appears to be a poor indicator. Linked to rare habitat type (mineral sites), but is described as being common in the region. Impacted by significant ecosystem changes from anthropogenic causes however, other indicators highlight impacts more distinctly.
	Christmas bird counts	3	6	4	May be a good indicator although data coverage in Puget Sound is limited. Also, evidence that indicator responds to management actions or pressures is lacking.
	Marbled murrelet presence at occupied sites	4	0	0	Overall good theoretical indicator. WDFW has monitored abundance, but apparent reliance on at-sea monitoring has made them harder to track. Threatened species with sensitivity to habitat fragmentation, a particular development concern in Puget Sound.
	Great blue heron	0	3	1	Do not have enough peer-reviewed evidence to support their use as an indicator. However, sensitive to development disturbance so may be useful in assessing landscape changes.
	Cavity nesting birds status & trends	4	4	1	Overall good indicator; reflect important functional guild and indicate significant land cover change impacts on species. Historical data trends lacking.
	Terrestrial bird species	3	6	4	May be good indicator but link to management is missing. Good data availability; migration timing may serve as leading indicator of climate change impacts.
Insects	Taylor's checkerspot butterfly status & trends	3	4	0	May be a good indicator although difficult to attribute population declines to human pressures (thought to be due to habitat loss). Not much good data available (appears mostly anecdotal).
	Insect species	Not yet evaluated			
Plants	Upland plant species	2	0	2	Theoretically-sound and responds to ecosystem attributes; data coverage is limited. Good indicator of species with relevance to ecosystem structure and function; may be anticipatory indicator through shifts in phenology.
	Terrestrial plant species status & trends	0	0	0	Poor indicator. Upland plant species is more targeted and more relevant.

Population condition. One indicator (Avian flu) has been identified for this attribute but has yet to be evaluated (Table 9). New indicators that characterize population condition of focal species should be developed for this section.

Table 9. Summary of Terrestrial Species - Population Condition indicator evaluations.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Birds	Avian flu	Not yet evaluated			

Interface species indicator evaluation

Population size. There were two indicators identified for interface species population size (Table 10). These indicators, stillwater breeding amphibians and amphibian and reptile species, have yet to be evaluated. Additional indicators that assess population abundance of focal species should be developed for this section.

Table 10. Summary of Interface Species - Population size indicator evaluation.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Amphibians & Reptiles	Stillwater breeding amphibians	Not yet evaluated			
	Amphibian & reptile species	Not yet evaluated			

Population condition. No indicators have been identified for interface species population condition. Candidate indicators may need to be developed for interface focal species population condition.

Food Web Evaluation

This version of the Puget Sound Science Update provides an initial evaluation of food web indicators, but is not intended to be comprehensive. Highlights include the evaluation of individual species or species complexes as food web indicators due to their key functional roles (e.g., forage fish, jellyfish), and the identification of existing data sources for assessing food web structure and function at Washington State agencies and via satellite.

Next Step: Future versions of this document would benefit from the evaluation of more indicators pertinent to the Freshwater and Terrestrial Domains, and the inclusion of more candidate indicators in the Marine Domain to ensure a full treatment of the key attributes identified in Section 3.2.3.3. Indicators of energy and material flows deserve particular attention in future assessments, as they were not the focus of the review by O'Neill et al. [34].

Key Point: Because of the difficulty of directly measuring attributes of food web health, ecosystem models have the potential to greatly contribute to the evaluation of foodweb indicators [118]. We encourage the development of ecosystem models as a tool for testing the performance of food web indicators.

Indicators of community composition

Species abundance and biomass data can be used to paint a synthetic picture of community composition, especially when viewed collectively with respect to particular Domains and in relation to species' trophic and functional roles. Even in isolation, insight into the status and trends of keystone species (i.e., those that have a disproportionate influence on food web structure relative to their abundance), highly connected species (i.e., those that are consumers of and consumed by many other species), minimally connected species, and those species representing a large proportion of the biomass in Puget Sound can be useful for interpreting the structural configuration of the food web [47]. In addition, species abundance and biomass data can be summarized into index values that describe the three different types of diversity defined in Section 3.2.3.3 (species, trophic, and response diversity). Dietary composition data, especially for higher trophic level predators such as marine mammals and birds, offer an alternative inroad to understanding community composition in Puget Sound and are available for a limited subset of species. Ongoing monitoring programs led by the Washington Departments of Ecology, Fish and Wildlife, and Natural Resources, among others, provide a rich source of information on community composition in Puget Sound. The challenge is to sort through these data to extract meaningful summary descriptions.

Indicators of energy and material flows

Proxies for primary productivity such as chlorophyll a concentration and phytoplankton biomass (in the Marine Domain) and leaf area index (in the Terrestrial Domain) are the most widely available indicators for energy and material flows in Puget Sound. Remote-sensing data are a valuable source of this information, though other, labor-intensive approaches are available for obtaining spatially explicit and finely resolved understanding of primary productivity as well (e.g., plankton tows, forest inventories, etc.). Alternatives to remote-sensing data are especially important in the Marine Domain, where it is difficult to obtain reliable estimates of primary productivity in nearshore areas at small spatial scales. More detailed data collection or modeling efforts (e.g., Ecopath with Ecosim) are needed to estimate the magnitude of secondary production and pathways of energy flows throughout the food web. Biogeochemical approaches for measuring cycling rates are well developed, especially with respect to inorganic nutrients, but such data are not widely available and can be quite expensive to obtain. Making up for this deficiency will require detailed, broad-scale studies of how different species interact with the physical and chemical oceanography of Puget Sound to affect processes such as nitrogen fixation, carbon sequestration, and microbial decomposition.

Evaluation of food web indicators in Puget Sound

There were nineteen Food Web indicators identified and of these, we have evaluated fifteen. The degree to which food web indicators satisfy our evaluation criteria is highly variable, and about half of them did not perform well against any of the criteria. The majority of evaluated indicators were from the Marine Domain, and no indicators have yet been evaluated for Freshwater Food Webs. The current status of indicator evaluations for the Food Webs Goal is summarized below.

Marine food web indicator evaluation

Eleven indicators of Marine Food Web community composition and two indicators of Marine Food Web Energy and Material Flows were evaluated (Table 11 and Table 12). The status and trends of benthic and pelagic fish communities species, marine shorebird diets, and jellyfish abundance performed best against the primary considerations for indicators of community composition. Of these indicators, however, only marine shorebird diets also met a majority of the Data and Other Considerations criteria. The general deficiency of quantitative data suggests the potential utility of several indicators while highlighting the need to begin data collection and monitoring. Most of the community composition indicators that did not perform well against the Primary Considerations also were deficient under the Data Considerations criteria. One of the biggest challenges for developing Marine Food Web indicators will be to increase their specificity prior to evaluation; several indicators, like the marine biodiversity index, shellfish, and benthic macroinvertebrates, were considered too vague to evaluate properly.

Phytoplankton biomass and chlorophyll a concentration provide similar information about primary productivity in the Puget Sound Marine Food Web. Both indicators performed well against the Primary Considerations for indicators of energy and material flows. However, chlorophyll a concentration met more of the Data and Other Considerations. Due to this indicator's reliance on remotely sensed data, however, it is unlikely to provide information about energy and material flows on spatial scales smaller than the PSP Action Areas. We suggest the evaluation of additional indicators of energy and material flows in the future.

Table 11. Summary of Marine Food Webs – Community Composition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Macro benthic inverts has peer-reviewed literature supporting 0 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Mammals	Harbor seals – food web interaction	1	5	2	Should be a good indicator of fish community composition, and possibly of population condition. Breadth of seal diet may limit power to detect small changes. Spotty historical data available throughout the region.
Key Fish	Benthic fish species status & trends	4	3	0	Overall appear to be good indicators of food web community composition, although historical data is currently lacking making it difficult to determine long-term trends.
	Bentho-pelagic fish status & trends	4	2	0	Overall appear to be good indicators of food web community composition, although historical data is currently lacking making it difficult to determine long-term trends.
	Bottomfish species (rats & flats) status & trends	1	0	4	Bottomfish noted as best indicator for some ecosystem attributes [118], although only appears as adequate indicator using our criteria. Difficult to determine if this indicator responds predictably to ecosystem attribute or actions/pressures. Patchy historical data.
Birds	Marine shore birds – food web interaction	3	7	2	Overall a good indicator, with relevance to forage fish prey species (diet variability responds to prey variability). Historical data available, although limited to two PSP action areas.
Shellfish & Other Inverts	Jellyfish	4	3	2	Theoretically-sound – jellyfish should be reliable indicators of trophic energy transfer & community composition. Responds predictably to actions and pressures, and may be especially relevant to understanding the status of forage fish. Historical data is limited, although still a promising indicator.
	Shellfish	0	0	0	Currently unable to properly evaluate because indicator is too vague. Recommend selection of particular species of bivalves as indicators.
	Macro benthic inverts	0	0	0	Currently unable to properly evaluate because indicator is too vague. Recommend selection of particular species of benthic inverts as indicators.
Key Species	Marine biodiversity index	0	0	0	Currently there is not sufficient information available to evaluate this indicator; the WA Biodiversity Council has planned to develop this indicator further.
	Marine fish/invert status & trends in marine reserves	0	0	0	Consolidate this indicator with 'marine fish/invert status & trends at rocky habitats'. If monitored inside marine reserve, it should also be monitored outside reserve to serve as a reference point.
	Marine fish/invert status & trends at rocky habitats	1	4	0	Difficult to evaluate as currently defined; need to explicitly define species or community parameters of interest. Some historical data available.

Table 12. Summary of Marine Food Webs – Energy and Material Flow indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example,

Chlorophyll a has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Plants	Phytoplankton biomass	3	1	1	Good indicator of pelagic ecosystems, especially nutrient cycling and the amount of primary production. Only limited amounts of historical data available. Provides similar information as chl a so choose one to avoid redundancy.
	Chlorophyll a	3	7	2	Chl a is a good proxy for overall primary productivity and nutrient cycling/uptake. Good historical data available. Phytoplankton biomass provides similar information to chl a concentration so choose one to avoid redundancy.

Freshwater food web indicator evaluation

Three indicators of Freshwater Food Web community composition were identified (Table 13), but unfortunately none were evaluated for this version of the PSSU. No indicators of Freshwater Food Web energy and material flows appear on the list of candidates suggested by O'Neill et al. [34]. Indicators of this Focal Component clearly deserve greater attention in future evaluation processes.

Table 13. Summary of Freshwater Food Webs – Community Composition indicator evaluations.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Key Fish	Freshwater fish biomass/stream length	Not yet evaluated			
Shellfish & Other Inverts	Macro invert assemblages – observed/expected	Not yet evaluated			
Key Species	Freshwater biodiversity index	Not yet evaluated			

Terrestrial food web indicator evaluation

O'Neill et al. identified one indicator of Terrestrial Food Web community composition (Table 14), the terrestrial biodiversity index [34]. Unfortunately, because it is still in development, this indicator did not meet many of the evaluation criteria under the Primary, Data, and Other Considerations. No indicators of Terrestrial Food Web energy and material flows were proposed by O'Neill et al. [34] and none were evaluated. As with Freshwater Food Webs, indicators of Terrestrial Food Webs clearly deserve greater attention in future evaluation processes.

Table 14. Summary of Terrestrial Food Webs – Community Composition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Terrestrial biodiversity index has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria.

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Key Species	Terrestrial biodiversity index	2	1	1	Fairly recent indicator developed by the WA Biodiversity Indicators Project. May be a good indicator for management, but more vetting required before fully usable for biodiversity assessment.

Interface food web indicator evaluation

Two related indicators of Interface Food Web community composition were identified by O'Neill et al. [34] (Table 15): forage fish and herring status and trends. Both indicators performed well against the Primary Considerations, though many of the Data and Other Considerations were not met. No indicators of Interface Food Web energy and material flows were proposed by O'Neill et al. [34] and none were evaluated. In general, new, additional indicators of this Focal Component should be evaluated in the future.

Table 15. Summary of Interface Food Webs – Community Composition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Forage fish status & trends has peer-reviewed literature supporting 4 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Guild	Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Key Fish	Forage fish status & trends	4	1	0	Theoretically-sound and relevant, but difficult to determine whether forage fish populations are responding to management actions or pressures or environmental conditions. Highly sensitive to uncontrollable environmental conditions.
	Pacific herring status & trends	4	1	0	Theoretically-sound and relevant, but difficult to determine whether forage fish populations are responding to management actions or pressures or environmental conditions. Highly sensitive to uncontrollable environmental conditions. Good data for many Puget Sound stocks.

Habitat Evaluation

This version of the Puget Sound Science Update provides an initial evaluation of habitat indicators, but is not intended to be comprehensive. Highlights include evaluation of marine and interface habitats (area and condition), as well as evaluation of a number of indicators of freshwater and terrestrial habitats condition. Many measures of habitat condition, especially those relating to water quality, were addressed under the PSP Water Quality goal.

- The inclusion of more candidate indicators for habitat area and pattern/structure (of all domains)
- Evaluation of habitat area and pattern/structure indicators for freshwater and terrestrial habitats
- Evaluation of freshwater habitats condition indicators
- Defining or identifying 'priority habitats' for priority habitats condition indicator (which appears under marine, freshwater, and terrestrial domains)

Commonly used data sources to evaluate habitat indicators included: Washington Departments of Ecology, Fish and Wildlife, and Natural Resources, and the Washington Biodiversity Council.

Indicators of habitat area and pattern/structure

Habitat area and pattern/structure are key measures of the overall health of a system, especially when they represent priority habitats. Insight into the status and trends of priority habitats area or pattern/structure can be used to infer changes in the status and trends of biota as well as abiotic processes. For example, changes in habitat area or pattern/structure can influence the amount of water runoff or coastal flooding, as well as regional species persistence. Thus insight into the status and trends of habitat area and pattern/structure can be useful for interpreting changes in ecosystem structure, function and processes.

Habitat area reflects the areal extent of a habitat as well as its shape, and can influence local population persistence and size for a single species [121]. While habitat area is important for maintaining biota, pattern/structure measures (e.g., the number of patches of each habitat, fractal dimension, and connectivity) also plays a significant role. The number of patches of each habitat (i.e., patch richness) may be correlated with species richness, thus monitoring patch number may be used to interpret trends in species biodiversity. Fractal dimension provides a measure of habitat complexity; natural areas tend to be more complex compared with human-altered areas, leading to changes in species richness [122, 123]. Connectivity between habitat patches affects the ability of an organism to cross between patches, and can be important for regional population abundance and survival [121]. WDNR monitoring programs, among others, provide an abundant source of information on habitat area in Puget Sound.

Indicators of habitat condition

Whereas the preceding attribute is concerned with measures of habitat area and pattern, it is also important to assess habitat quality or condition. Habitat condition reflects the basic needs of a species (e.g., food, water, cover) and is a critical component to predict species distributions [42] and population abundance and survival [121]. For example, important variables for fish habitat would include water quality parameters (e.g., DO levels, temperature) as well as the presence and abundance of non-native invasive species or nuisance species that compete for resources. Thus, habitat condition refers to abiotic (i.e., physical and chemical properties) and biotic properties (e.g., invasive or nuisance species, dominant species), as well as dynamic structural characteristics.

Abiotic properties (e.g., water and benthic quality parameters) are the most widely available indicators for habitat condition in Puget Sound. However, according to the PSSU framework, they fall under the Water Quality goal and will therefore be discussed in that section. Biotic properties, such as the status and trends of harmful algal blooms or the presence of nuisance species, are a key measure of habitat health and can be used to interpret changes in native species abundance, distribution, and survival. Dynamic structural characteristics cause changes in physical habitat complexity and morphology, and are included in habitat condition because they maintain (or eliminate) the diversity of natural habitats. Data collection led by WDNR, WDFW, and the Washington Biodiversity Council provides important information on habitat condition in Puget Sound.

Evaluation of habitat indicators in Puget Sound

There were sixty habitat indicators identified by O'Neill et al. [34] and of these, we have evaluated thirty-seven. The majority of those evaluated are indicators of area and condition for marine and interface habitats. A small subset of indicators has been evaluated for Freshwater and Terrestrial Habitats, and future versions of this document should focus on completing these evaluations. The current status of indicator evaluations for each habitat focal component is summarized below.

Marine habitat indicator evaluation

Area and Pattern/Structure. Three indicators of marine habitat area were identified (Table 16). Of these, two (eelgrass status and trends and kelp status and trends) were evaluated and performed adequately against the criteria. Both indicators were theoretically-sound, but do not respond predictably to management actions or pressures. In particular, it is difficult to determine causes of variation in habitat area (e.g., natural vs. anthropogenic impacts). Ongoing monitoring programs led by WDNR and the Pacific Northwest National Laboratory, among others, provided extensive information for these indicator evaluations.

Table 16. Summary of Marine Habitats – Area and Pattern/Structure indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Eelgrass status & trends has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Eelgrass status & trends	2	4	2	Theoretically-sound but difficult to determine what causes changes in abundance (natural vs. anthropogenic).
Kelp status & trends	2	5	3	Theoretically-sound but response is limited to floating kelp. Difficult to determine causes of variation in abundance (especially indirect impacts).
Marine macro algae	Not yet evaluated			

Habitat Condition. There were seventeen indicators of marine habitat condition identified (Table 17). The majority of those listed refer to biotic properties (e.g., non-native invasive aquatic species); considerably fewer relate to abiotic properties. Two indicators (upwelling zones and marine water quality parameters) were evaluated under Marine Water Quality; three indicators (non-native invasive marine species threat, number of marine native nuisance species, and priority habitats condition) have yet to be evaluated. Several indicators performed poorly against all criteria because we were unable to determine what they were an indicator of. These included the number of salmon net pens, number of oyster culture sites, and number of clam culture sites, and may better serve as 'pressure' indicators.

Table 17. Summary of Marine Habitats – Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Non-native invasive aquatic marine species has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Upwelling zones	Evaluated under Marine Water Quality			
Aggregation/deposition zones	3	5	1	Theoretically-sound. Could be a good leading indicator of habitat forming processes.
Marine water quality parameters	Evaluated under Marine Water Quality			
Harmful algal blooms (HABs) status & trends	3	7	2	Good indicator of habitat condition, but does not respond predictably to management actions or pressures b/c lack of understanding of the conditions for HAB formation. Monitoring needs to be spatially and temporally explicit b/c no way to forecast HABs more than 1-2 wks out; this increases costs.
Intertidal biotic community status & trends	0	4	0	Currently unable to find sufficient evidence supporting the use of this indicator.
Non-native invasive aquatic marine species	2	3	3	Possibly theoretically-sound. Lacking evidence explicitly linking presence/absence to changes in habitat condition. Some existing data in Puget Sound. Most useful if continuous monitoring for presence/absence throughout the Sound.
Non-native invasive marine species threat	Not yet evaluated			
Number of marine native nuisance species	Not yet evaluated			
Number of salmon net pens	0	0	0	Unable to determine what this is an indicator of – may better serve as a 'pressure' indicator.
Number of oyster culture sites	0	0	0	Unable to determine what this is an indicator of – habitat condition, water quality, or human health?
Number of clam culture sites	0	0	0	Unable to determine what this is an indicator of – habitat condition, water quality, or human health?
Priority habitats condition	Not yet evaluated			
Number of marine species at risk that are threatened/endangered /candidate	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of marine species listed under Federal ESA	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of marine species of concern on State list	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of marine species of greatest conservation need	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of marine species of conservation concern	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.

A subset of related indicators performed well against all criteria and included the number of marine species at risk that are threatened/endangered/candidate, number of marine species listed under Federal ESA, number of marine species of concern on State list, number of marine species of greatest conservation need, and number of marine species of conservation concern. These indicators were originally evaluated under Marine Species (population condition), but were moved to Marine Habitats because the absolute number of species on any of these lists is a better reflection of habitat or environmental condition. All were theoretically-sound, but because each indicator is a compilation of species, it is difficult to conclude whether they respond predictably to management actions. These indicators appear to convey redundant information. Information on these indicators was principally obtained through WDFW and the Washington Biodiversity Council.

Two indicators, aggregation/deposition zones and harmful algal blooms status and trends, performed well against primary and data considerations. The remaining indicators (intertidal biotic community status and trends and non-native invasive aquatic marine species) received poor evaluations. Monitoring efforts by WDFW, WDOH, WDNR, among others, provided important data sources for these evaluations.

Freshwater habitat indicator evaluation

Area and Pattern/Structure. O'Neill et al. (2008) identified three indicators for freshwater habitat area (Table 18) [34]. These indicators (freshwater physical habitat, floodplain connectivity, and instream habitat) have yet to be evaluated. As well as evaluating these indicators, it may be useful to develop additional candidate indicators for this section.

Table 18. Summary of Freshwater Habitats – Area and Pattern/Structure indicator evaluations.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Freshwater physical habitat	Not yet evaluated			
Floodplain connectivity	Not yet evaluated			
Instream habitat	Not yet evaluated			

Habitat Condition. Eighteen indicators of freshwater habitat condition were identified, half of which have not been evaluated (Table 19). Several indicators including max temperature, sediment loadings rate, stream and lake water quality parameters, and spawning habitat water quality, are evaluated under Water Quality though they do pertain to habitat condition.

Table 19. Summary of Freshwater Habitats – Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Number of freshwater species of conservation concern has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Max temperature	Evaluated under Freshwater Quality			
Sediment loadings rate	Evaluated under Freshwater Quality			
Number of fish barriers corrected	Not yet evaluated			
Percent of channel length armored	Not yet evaluated			
Number of artificial fish barriers	Not yet evaluated			
Stream water quality parameters	Evaluated under Freshwater Quality			
Lake water quality parameters	Evaluated under Freshwater Quality			
Spawning habitat water quality	Evaluated under Freshwater Quality			
Non-native invasive aquatic species threat	Not yet evaluated			
Number of freshwater native nuisance species	Not yet evaluated			
Non-native aquatic freshwater species	Not yet evaluated			
Priority habitats condition	Not yet evaluated			
Clean & cool water for salmon	Not yet evaluated			
Freshwater physical habitat condition	Not yet evaluated			
Number of freshwater species listed under the Federal ESA	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of freshwater species of concern on State list	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of freshwater species of greatest conservation need	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of freshwater species of conservation concern	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.

Evaluated indicators for freshwater habitat condition represent a group of related indicators that performed well against all criteria. These included the number of freshwater species listed under Federal ESA, number of freshwater species of concern on State list, number of freshwater species of greatest conservation need, and number of freshwater species of conservation concern. These indicators were originally evaluated under Freshwater Species (population condition), but were moved to Freshwater Habitats because the absolute number of species on any of these lists better reflects habitat or environmental condition. All were theoretically-sound, but because each indicator is a compilation of species, it is difficult to conclude whether they respond predictably to management actions. These indicators appear to convey redundant information. Information on these indicators was principally obtained through WDFW and the Washington Biodiversity Council.

Terrestrial habitat indicator evaluation

Area and Pattern/Structure. O'Neill et al. (2008) identified three indicators of terrestrial habitat area: terrestrial land cover status and trends, transportation impacts, and forests and forestry (Table 20) [34]. None of these indicators have been evaluated. This section may benefit from the addition of new candidate indicators, as well as evaluating the indicators currently identified.

Table 20. Summary of Terrestrial Habitats – Area and Pattern/Structure indicator evaluations.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Terrestrial land cover status & trends	Not yet evaluated			
Transportation impacts	Not yet evaluated			
Forests & forestry	Not yet evaluated			

Habitat Condition. There were nine indicators of terrestrial habitat condition identified (Table 21). Three indicators, old growth forest change, road densities, and priority habitats condition, have yet to be evaluated. Two indicators, non-native invasive terrestrial species threat and number of terrestrial native nuisance species, performed well against primary considerations but not data considerations. The Washington Invasive Species Council is leading efforts to compile numbers and occurrence data for these two indicators.

Table 21. Summary of Terrestrial Habitats – Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Number of terrestrial species of conservation concern has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Old growth forest change	Not yet evaluated			
Road densities – erosion	Not yet evaluated			
Non-native invasive terrestrial species threat	3	1	2	Theoretically-sound, but little data currently exists. WA Invasive Species Council leading efforts to compile numbers and occurrence data.
Number of terrestrial native nuisance species	3	1	2	Theoretically-sound, but little data currently exists. WA Invasive Species Council leading efforts to compile numbers and occurrence data.
Priority habitats condition	Not yet evaluated			
Number of terrestrial species listed under Federal ESA	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of terrestrial species of concern on State list	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of terrestrial species of greatest conservation need	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.
Number of terrestrial species of conservation concern	3	7	4	Theoretically-sound, although peer-reviewed evidence linking this with habitat condition is lacking. Difficult to say how this indicator responds predictably to management actions or pressures b/c it is a compilation of species. May be a good vital sign indicator. To avoid redundancy, choose one indicator of species conservation concern.

A subset of related indicators performed well against all criteria and included the number of terrestrial species listed under Federal ESA, number of terrestrial species of concern on State list, number of terrestrial species of greatest conservation need, and number of terrestrial species of conservation concern. These indicators were originally evaluated under Terrestrial Species (population condition), but were moved to Terrestrial Habitats because the absolute number of species on any of these lists better reflects habitat or environmental condition. All were theoretically-sound, but because each indicator is a compilation of species, it is difficult to conclude whether they respond predictably to management actions. These indicators appear to convey redundant information. Information on these indicators was principally obtained through WDFW and the Washington Biodiversity Council.

Interface habitat indicator evaluation

Area and Pattern/Structure. There were four indicators identified for interface habitat area (Table 22). Wetland acreage status and trends has not been evaluated. Two indicators, saltmarsh status and trends and riparian habitat, performed well against all criteria. In particular, riparian habitat fulfilled all of the primary considerations as well as most of the data considerations. Of note, saltmarsh status and trends did not fulfill the theoretically-sound criteria because it is most often used as part of an integrative assessment of ecosystem health, rather than a stand-alone indicator. Shoreline geomorphology received a poor evaluation because, while it is theoretically-sound and relevant to management, data trends are largely missing, especially as they relate to changes from natural versus anthropogenic impacts. Monitoring efforts by WDNR and Simenstad et al. [124] provided valuable data for these evaluations.

Table 22. Summary of Interface Habitats – Area and Pattern/Structure indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Saltmarsh status and trends has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Wetland acreage status & trends	Not yet evaluated			
Saltmarsh status & trends	3	7	4	Overall good indicator -- total area often used as integrative assessment of ecosystem health. May best be used as part of an integrative assessment of habitat change in the region.
Riparian habitat	5	6	3	Very good indicator of riparian ecosystem health including habitats and species. Evidence that restoration increases riparian habitat area. Good data for Puget Sound. May best be used as part of an integrative assessment of habitat change in the region.
Shoreline geomorphology	2	0	0	Poor indicator. While this indicator is theoretically-sound and relevant to management, it fails all other criteria. Indicator requires classification of shorelines, which groups throughout PS do differently. Also, difficult to determine (1) when one geomorphic type ends and another begins, and (2) natural vs. anthropogenic change.

Habitat Condition. Percent of shoreline armored, nearshore physical and biotic habitats, and wildlife status and trends in restored habitats were selected as indicators for interface habitat condition (Table 23). All were theoretically-sound and relevant to management. Percent of shoreline armored may be a good indicator, although explicit linkages between armoring and effects on biota is largely absent. Nearshore habitats met most of the data and other considerations, and may be useful as a leading indicator of how habitat-forming processes have been altered in the nearshore environment. Wildlife status in restored habitats appears to be costly and time intensive to measure. Principal data sources for these evaluations included monitoring efforts by WDNR, as well as Simenstad et al. [124].

Table 23. Summary of Interface Habitats – Condition indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Nearshore physical and biotic habitats has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Percent of shoreline armored	3	5	2	May be a good indicator, although there is not a lot of science concerning how this affects biota (i.e., difficult to determine whether it responds predictably to ecosystem attributes). Also, difficult to determine thresholds – how much armoring in an area is bad? Easily measured, and cumulative effects important especially in the context of other shoreline stressors.
Nearshore physical & biotic habitats	2	5	4	Theoretically-sound, although few studies relating shoreform change to nearshore ecological function. Primarily useful as a leading indicator of how habitat forming processes have been altered in nearshore (i.e., measures level of impairment to habitat forming processes).
Wildlife status & trends in restored habitats	2	2	0	Good measure of restored habitat's ecological function, but useful measures (growth, consumption, survival) rather than number and diversity are more costly and time intensive to measure. Data rarely available.

Water Quality Evaluation

Recently the PSP listed several contaminants of concern for Puget Sound organized into four general categories including toxics, nutrients, pathogens, and other (i.e. deviations in physical/chemical state of a water body; [125]). Specific issues related to these categories, including discussions on several chemicals of concern, have been detailed therein and elsewhere [76]. Nutrients and "other," will be discussed as physical/chemical parameters; toxics as trace inorganic and organic chemicals; pathogens, under the goal Human Health. ''' 5.4.1 Indicators of Hydrodynamics'''

Water circulation patterns in Puget Sound influence water quality. Freshwater inputs from rives and streams can create density stratification, which, in turn, can exacerbate conditions underlying eutrophication and hypoxia [126]. Washington State Department of Ecology reports on stratification based on frequency and intensity. Stratification intensity is based on change of seawater density (reported a sigma-t; density in kg m-3 – 1000) over the pycnocline. Frequency is determined by the percent of time that the change in density across the pycnocline is greater than two. Stratification patterns vary temporally and locally within Puget Sound; stratification is generally strongest near areas of freshwater inflow while vertical mixing occurs at sills [90]. Status and trends of stratification are discussed in the sections on hypoxia and marine eutrophication in Chapter 2 of the Puget Sound Science Update.

Marine circulation may be the largest factor in the delivery of nutrients to Puget Sound [86]. Periodic deep water intrusions over the entrance sill at Admiralty Inlet deliver marine waters into Puget Sound [88]. Deep water circulation and residence times vary throughout Puget Sound, and also interannually; interannual variations appear to be associated with variations in freshwater flows, and salinity at the Strait of Juan de Fuca [127, 128]. Large sale climate variations can affect upwelling off the Strait of San Juan de Fuca (and, thus, salinity), surface winds, temperatures, and precipitation, possibly influence Puget Sound's oceanography [89, 129]. Wind may be important driver on the circulation of Puget Sound. Wind has been implicated in causing outcrops of low-DO water in southern Hood Canal [88].

Although marine circulation patterns are likely important, particularly in terms of nutrient supply to Puget Sound, the magnitude, timing, and influencing factors are not well understood.

Indicators of Physical/Chemical Parameters

Physical and chemical parameters can define the state and status of water with regard to the health of humans and the environment. These include temperature (T), dissolved oxygen (DO), nutrients such as nitrogen (N) and phosphorus (P), chlorophyll, and the Secchi depth. These fundamental measures are often combined into various indices or states, depending on management concerns.

Low DO is of particular concern in marine waters, particularly in the Hood Canal and areas of South Puget Sound [76]. A discussion of the status and trends is included in Chapter 2 of this Puget Sound Science Update. A discussion of the potential biological effects of low DO are included in a literature review performed by the Washington State Department of Ecology as part of an evaluation of DO standards for marine and freshwaters [92, 130, 131]. A brief discussion of the DO standards is presented in Section 6.8.3.

Temperature is a critical measure and of importance to instream biota in streams and rivers of the region. A discussion of the biological impacts of temperature is included in the literature review performed by the Washington State Department of Ecology ([93]; see Section 6.8.3). There is currently limited evidence that temperature changes are important in the marine environment of Puget Sound.

Eutrophication, nutrients, chlorophyll, and Secchi depth are measures related to the productivity of a water body [86, 95-101, 132-135]. Marine eutrophication is discussed in Chapter 2 of this Puget Sound Science Update. An evaluation of the water quality criteria for phosphorus and its relationship to Secchi and trophic state has been performed by the Washington State Department of Ecology [136].

The Washington State Department of Ecology and King County utilize a freshwater Water Quality Index (WQI) to summarize water quality information in a format that is easily understood [137]. The WQI is based on T, DO, pH, fecal coliform bacteria (FC), TN, TP, total suspended sediment (TSS), and turbidity. Ranking factors are based on relations to state water quality standards (T, DO, pH, and FC; [138]), the limiting nutrient (TN or TP) or a calculated harmonic mean (TSS and turbidity). Evaluations of the WQI approach suggest that it be a communication tool (e.g. a reporting indicator) but not used for evaluation (e.g., an assessment indicator) since it does not reveal specific water quality traits [137, 139-141]. It has also been suggested that subjective, professional judgment be minimized in the development of WQIs by using published cause/effect relationships [142].

Rivers and streams in Canada utilize a Canadian WQI (CCME WQI) that is similar to the WQI developed by Washington State Department of Ecology. However the CCME WQI reflects Canadian standards and is adjusted by the scope, frequency, and amplitude of failed test values [143].

Marine WQIs are currently not used in the Puget Sound region, though one is under development. Washington State Department of Ecology has reported on areas where water quality is a concern by summing the results of five water quality indicators (stratification, DO, nutrients, FC, and ammonium; [126]).

Indicators of Trace Inorganic and Organic Chemicals

The marine waters and sediments of Puget Sound have been affected by different classes of anthropogenic chemicals (e.g. toxics); some have been well studied while others less so. Several efforts have been made to identify the chemicals of concern in Puget Sound based on historic monitoring programs [144-146]. These toxic chemicals included metals and metalloids (arsenic, cadmium, copper, lead, mercury, and tributyl tin), organic compounds (polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), pesticides, dioxins and furans, phthalate esters, polybrominated diphenyl ethers (PBDEs), and hormone disrupting chemicals. In 2007, the Washington State Department of Ecology, as part of a Chemicals of Concern work group, modified this list resulting in the following 17 chemicals of concern for marine waters [145].

- Arsenic
- Cadmium
- Copper
- Lead
- Mercury
- Total PCBs
- Low molecular weight PAHs
- Carcinogenic PAHs
- Other high molecular weight PAHs
- DDT and Metabolites
- Triclopyr
- Dioxins and furans
- bis(2-Ethylhexyl)phthalate Phthalate esters
- Total PBDEs
- Nonylphenol
- Oil or petroleum product
- Zinc

Subsequent evaluations added other broad categories of toxics including pharmaceuticals and personal care products [76, 147]. These are of concern because of their observed or presumed ability to cause harm to human health or the environment.

There are several state and local monitoring efforts, which address many of these chemicals of concern. Chapter 2 of this Puget Sound Science Update reviews the status and trends of Persistent Bioaccumulative Toxics (PBTs), which includes PCBs, PDBE, pesticides (i.e. DDT) and mercury, PAHs, metals, and endocrine disrupting chemicals.

The prioritization of toxics in water and sediments for monitoring, evaluation, and potential remediation is complex and difficult, particularly considering the vast array of emerging contaminants in aquatic environments [148]. In order to determine whether a compound is of concern it is necessary to understand its source, distribution, fate and transport, exposure and biotic effect. And, although a significant amount is known about certain toxics, very little is known about the majority of them [149]. The USGS performed a national reconnaissance, sampling in 139 streams and analyzing for 95 toxics and found a common detection of multiple contaminants in each sample [150]. Further sampling programs have been performed for groundwater and untreated drinking water sources [151, 152]. Similar suites of chemicals were found in the groundwater and untreated drinking water sources compared to the river and streams, though at a lower detection frequency and generally lower concentrations. Similar results have been reported for European sampling surveys [153, 154].

King County performed a preliminary survey of sixteen known endocrine disrupting chemicals in marine waters, lakes, rivers, and small streams [155]. Overall levels were similar to those found in national surveys. Specific compounds such as 17α-ethinylestradiol (EE2) and 4-nonylphenol were detected frequently and at maximum levels greater than the effective concentrations reported in the literature.

Emerging contaminants often occur at very low concentrations and in mixtures; accurate risk assessments may depend on the use of relevant exposure scenarios to capture potential synergistic or antagonistic effects [156]. For example, individual estrogenic chemicals can act additively, causing a response even when the concentration of each individual compound is below the known effective concentration [157]. In addition to endocrine disruption, environmental estrogen exposure has been reported to induce genotoxic damage, affect immune function, and alter metabolism in fathead minnows, [158]. Further, responses to EE2 may be different with mixtures of endocrine disruptors compared to EE2 alone, suggesting complex interactions.

This suggests that emerging contaminants are present in Puget Sound and may be environmentally significant. As such, indicators of water quality related to these trace inorganic and organic chemicals should be evaluated and selected carefully. Sumpter and Johnson suggest two possible approaches to evaluate the potential risks and effects associated with emerging contaminants [159]. One would be to use contaminant-specific information to identify possible exposure-effect relationships combined with hydrology to identify potential hotspots and focus analytical investigations. The second approach would begin with investigations of biota directed in specific locations by hydrologic modeling to determine if there are any identifiable adverse impacts. Both investigatory approaches may be useful in evaluating relative threats from emerging contaminants as the relative threats are currently not known.

The analytical-chemical approach and biota-observation approach are both used for monitoring water quality and the selection/utilization of specific indicators. One issue specifically related to the selection and evaluation of water quality indicators is whether they are better suited as indicators of water-quality or of species condition (or, perhaps, are good indicators of both). The Heinz Foundation (2008) reports contaminants in fish in shellfish as a measure of chemical contamination of the environment where as EPA’s Science Advisory Board (2002) reports contaminants in tissue as a sign of disease potentially affecting species condition [42, 44]. For the purpose of this report we recognize contaminants in tissue (i.e. tissue residue levels) and biomarkers of contaminant effects as measures indirect indicators of water quality and direct measures of species condition, however species will vary in the ability to reflect local, regional and coastal water quality condition.

There are several indicators of contaminants in biota, which could be either measures of Water Quality – Trace Inorganic or Organic Chemicals, or Species – Population Condition. For example, the level of contaminants and/or liver disease in English sole has been shown to be strongly correlated with the level and presence of polycyclic aromatic hydrocarbons (PAH) in sediments, while also being a measure of species health [160-165]. This suggests that liver disease in English sole can be a suitable measure of general Marine Water Quality (i.e., PAHs in sediments) or of Species Population Condition.

Vitellogenin (Vtg) production in male fish may be another useful marker of environmental exposure to xenoestrogens [166] although unlike liver disease, the causative agent cannot be clearly identified. In Puget Sound, elevated levels of Vtg have been reported for English sole [167].

Recently, several studies investigating the causative action of xenoestrogens have implicated the disruption of steroidogenic acute regulatory (StAR) protein activity, which may be key in moderating the rate limiting step in steroid homone syntheses; evaluating StAR protein activity, then, may be a valuable biomarker for xenoestrogen exposure [168-170].

As these examples illustrate, the value of measuring biological response in biota (i.e. Vtg induction in male fish or liver disease in English sole) as an indicator or water quality is dependent largely on the strength of the knowledge of the exposure-effect relationship as well as the chemical specificity of the of the reaction. A lack of knowledge or a weak causal link would imply that the biological response were a poor indicator of water quality.

The concentration of specific contaminants in aquatic organisms may be appropriate indicators of water quality or species condition. Measurements of PAH, PCBs, PBDEs (and metals) and metabolites in fish tissues, primarily salmonids and bottom fish, and associated health effects, have been well studied in the region [171-175]. In some cases (i.e. PAHs, PCBs, and tributyl tin), the evaluation of tissue and sediment data have been used to establish sediments quality thresholds [164, 176, 177]. In other cases the presence of contaminants in biota may be reflective of environmental conditions, though health effects and thresholds are not well defined [178, 179].

The use of toxics in biota as indicators of water quality in Puget Sound is discussed below.

The NOAA National Status and Trends Mussel Watch Program has monitored contaminant concentrations in the coastal United States, including at least thirteen sites in Puget Sound, by sampling mussels, oysters, and sediments [180, 181]. Mussels have been shown to take up and accumulate the bioavaible fraction of hydrophobic contaminants from the water column [182]. Tissue concentrations of PAHs, total PCBs, and total DDTs were higher in mussels from the urban-associated sites compared to those from less urban areas; adverse health effects were observed [183, 184]. In Puget Sound, results indicated no significant trends at most sites, though several had decreasing trends and a few (Se) had increasing trends with time [180]. These results are discussed in Chapter 2 of this report. Toxics contaminants in mussels may be an appropriate indicator of water quality.

Tissue sampling of resident Pacific herring populations may allow for general indications of water quality. However, because herring populations range widely and feed on planktonic organisms (e.g., krill), their contaminant levels reflect conditions in the pelagic food web on a large, regional scale. West et al. (2008) was able to discriminate differences in contaminant levels between herring populations sampled from inner and outer Puget Sound (i.e. north and south of Admiralty Inlet) but not among inner Puget Sound populations [185].

Due to the lifecycle and migration traits, measures of toxics in adult salmonids may not be suitable as indicators of local or regional water quality [186]. It has been shown that over 98% of adult body mass of six Pacific salmon species and steelhead is acquired while feeding in marine waters [187] but populations of Pacific salmon among and within species vary considerably in their marine range and distribution. Adult Chinook salmon may accumulate over 95% of their persistent organic contaminant burden during their time at sea, with their final tissue contaminant concentrations reflecting the range of exposure throughout their marine water feeding areas [186, 188]. In contrast, recent work has suggested PCB concentration in tissues of localized outmigrating juvenile populations may be correlated with local sediment concentrations [189].

Tissue analysis of harbor seals in Puget Sound and Strait of Georgia found relatively high levels of PCBs, polychlorinated dibenzo-p-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs), and that location partially explained the relative concentrations and the mixture profiles [190]. Weight of evidence suggests that harbor species are exposed to levels of contaminants that have the potential to cause adverse health effects [188]. Although the range of harbor seals is relatively small they consume a wide-variety of fish, both local and ranging, suggesting that harbor seal contamination may be somewhat disconnected from that of their local habitats. As such, they may not be useful as indicators of localized sediments or water column contamination. However, a food basket analysis indicated that variances of contaminant concentrations in harbor seal population could serve as indicators of food web contamination, and environmental contamination on a regional scale [189].

Tissue samples from free-ranging killer whales found very high levels of PCBs and also of PCDDs and PCDFs [188]. The increasing presence of PDBEs in the killer whale food chain may also be of increasing import [191]. The range of the killer whales, and the range of their diets, suggests that tissue contaminant levels may not correlate well with local or regional contaminant conditions [185]. These reports suggest that there are measures of toxics in biota may be suitable measures of water quality at local (e.g., bivalves) and regional (e.g., herring, juvenile salmonids, or Harbor seals) though appropriate selection is necessary depending on the management concern. Toxics in biota can also be utilized as measures of species condition, though the health effect thresholds are not always clear for all species of concern.

Evaluation of Water Quality Indicators

Fifty-seven water quality indicators were selected for evaluation, and thirteen were evaluated. In general the indicators that were evaluated performed well against the Primary Considerations. However, there were often gaps in data, either spatially or temporally.

Marine Water Quality

A summary of the evaluation of indicators of Marine Water Quality is shown in Table 24. The indicators of marine water quality generally performed well against the criteria suggesting that there are many acceptable indicators, which can be selected depending on the issue of management concern. Generally, the indicators evaluated under Physical/Chemical parameters performed well under the Primary Considerations, and the Data Consideration. However, there were often limitations in the spatial and historical extent of the data.

Table 24. Summary of Marine Water Quality indicator evaluations. The numerical value under each consideration represents the number of evaluation criteria supported by peer-reviewed literature. For example, the indicator Toxics in Mussels has peer-reviewed literature supporting 4 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary
Marine Water Quality				
Hydrodynamics				
Seawater stratification				
Upwelling zones				
Flushing rates				
Marine Water Quality				
Physical/Chemical Parameters				
Benthic infaunal community structure (sediment quality)				
Marine water quality index				
Nutrients in marine waters	5	4	3	Very important in specific nutrient limited locations (e.g. Hood Canal, Budd Inlet) though less so in main body of sound as it is generally not nutrient limited. High nutrients can lead to eutrophication and associated effects = high management concern. Management actions can affect some sources of anthropogenic nutrients. Reference points and targets are site specific and depend on historical state of water body. Certain areas of concern such as Hood Canal and Budd Inlet have good and sufficient coverage, though other areas is limited.
Sensitivity to eutrophication	0			Eutrophication is not a good indicator in itself. Eutrophication is characterized by a suite of measures such as DO, HABs, nutrients which are other specific indicators. Phytoplankton biomass is measured elsewhere. "Sensitivity," is not readily measured. Eutrophication is not directly measured nor is sensitivity to eutrophication. Makes this unsuitable for an indicator.
DO marine	5	4	4	DO levels affect marine species. Selected areas of low DO in Puget Sound are of great management concern. Management actions may have some impact on anthropogenic nutrient inputs to PS. Generally clear reference points and targets though may vary depending on historic conditions. Some areas of localized coverage, though not good historical record.
Marine water quality parameters				
WWTP nutrient hot spots				
Ratio of point to non-point nutrient loads				
Marine Water Quality				
Trace Inorganic and Organic Chemicals				
Toxics in English sole	5	5	4	Contaminant levels in English sole (including PAH metabolites in bile) increase with concentrations in the environment. Some metals (e.g.,Hg) are more sensitive to increase in fish age. Some metals (e.g. Cu) are regulated by fish and therefore tissue residues of Cu are not very sensitive measures of water quality. For example, tissue resides of Cu in Puget Sound marine fish do not vary among species or among locations within a species. Defined thresholds exist for some chemicals. Measurement and evaluation requires specialized techniques and instrumentation. Historic coverage of over 50 sites but spatial coverage was reduced in 2001 to 8 sites, representing urban, near-urban and non-urban site. Need to accounts for variation in age and lipid content of fish.
Liver disease in English sole				Prevalence of liver disease (i.e.toxicopathic hepaocellar lesion) is elevated in PAH contaminated environments. Changes in prevalence of iver disease are used to document reductions PAH environmental contamination associated with management strategies to reduce source control and remediate sediments. Thresholds for PAH levels in sediment associated with increased prevalence have been defined. Data collection requires technical expertise. Historic coverage of over 50 sites but currently limited to 8 sites representing urban, near-urban and non-urban site. Need 60 fish per samples location and % prevalence must be statistically corrected to account for age in the fish.
Toxics in clams				DOH and King County completed studies in the mid-90s but discontinued sampling in part because of low number of detects for organic compounds and variability of metals data, possibly associated with inconsistent species being sampled.
Fecal pollution index for commercial shellfish beds				
Chemical contamination in Puget Sound sediments				
Abiotic/pollutant exposure condition				
Toxics in crabs & shrimp				
Toxics in adult Chinook and Coho salmon	4	6	4	Toxics in biota generally reflect contaminants in their environment. High variability of toxic conc, especially for Chinook salmon associated with fish's residency in Puget Sound; tissue residue will vary substantially with changes in residency which may mask changes local water quality. Elevated toxics in salmon are pertinent to PSP goals for water quality, human health and species and food webs. Reflects toxics in marine waters throughout salmon's marine distribution. Data coverage includes populations returning to Nooksack, Skagit, Duwamish, Nisqually, and Deschutes rivers. Sampling from 1991; Chinook salmon discontinued in 2006. There is a low signal-to-noise ratio as residency of fish is often unknown.
Toxics in harbor seals	3	6	3	Some variability in tissue concentrations associated with variation in diet among seals from different sampling sites; reflects regional water quality (i.e. Georgia Bas vs, Puget Sound). Effects thresholds are based on captivity studies. Limited number of sample locations published to date. Archived samples for PCB and PBDE temporal trends at one locations.
Toxics in Pacific herring	5	8	4	Reflects toxics in marine waters throughout herring's distribution. Elevated toxics in Pacific herring are pertinent to PSP goals for water quality, human health and species and food webs. Concentration differences between northern Puget Sound and central Puget Sound are detectable. Specific threshold for herring exist of PAHs but not other chemicals. Coverage for major Puget Sound basins from 1999; no temporal trends observed.
Toxics in mussels	4	5	4	Data for toxics in mussel in Puget Sound are collected as part of NOAA's national Mussel Watch program. Number of sites is limited especially in southern Puget Sound. Currently a non-random sampling design is used. Threshold specific to the health of mussels are not know.
Fecal bacteria in offshore Puget Sound				
Fish Tissue Contaminants Index				Whole body samples of fish analyzed for contaminants, therefore not suitable for human health. Some problems interpreting data as species, sizes and ages vary among locations. Possibly combine these data with other Puget Sound datasets (e.g. ENVEST and WDFW).
Toxics in Osprey eggs				Only 2 stations are sampled in Puget Sound.
Oil Spills				
PCBs in Cormorant eggs				Data exist for the St. Georgia but limited data is available for Puget Sound
Star protein/ DNA damage				moved to species condition
Vtg induction in male fish	3	3	4	Elevated levels Vtg indicate exposure to xenoestrogens, including some trace organics. Various biological effects have been have been correlated with magnitude of Vtg induction in male fish but threshold will vary by species. Broad spatial coverage for English sole in Puget Sound. Limited time series data (e.g. 2-3 yr)s at some site. Very sensitive to changes in xenoestrogen.

There are several indicators concerning measures of contaminants in ecological receptors, which could be either measures of Water Quality – Trace Inorganic or Organic Chemicals, or Species – Population Condition (see section 5.4.3). The initial indicator organization placed these indicator based on trophic level and management concern. Low-trophic-level species were considered to be more directly exposed to environmental contaminants and thus more representative than were higher-trophic-level species. Toxics in species with high management concern were placed under population condition. The detailed evaluation process allowed for reorganization, as appropriate.

Interface Water Quality

A summary of the evaluation of indicators of Marine Water Quality is shown in Table 25. To date, only one indicator has been evaluated against the criteria.

Table 25. Summary of Interface Water Quality indicator evaluations. The numerical value under each consideration represents the number of evaluation criteria supported by peer-reviewed literature. For example, the indicator Toxics in Juvenile Salmon has peer-reviewed literature supporting 5 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary
Interface Water Quality				
Hydrodynamics				
Aggregation/deposition zones				
Interface Water Quality				
Physical/Chemical Parameters				
Sediment Quality Triad Index				
Wetland Water Quality Index				
Nearshore water quality				
Wetland water quality				
Interface Water Quality				
Trace Inorganic and Organic Chemicals				
Pesticide poisonings in raptors				Limited date is available for Puget Sound; consistently measurable, responsive to change. Limited study was not maintained.
Toxics in heron eggs				
Toxics in Juvenile Salmon	5	5	4	A consistent monitoring program for toxics in juvenile salmon does not exist for Puget Sound, however, multiple studies complete date, meet most of the criteria used to screen indicators.

Freshwater Quality

A summary of the evaluation of indicators of Freshwater Quality in shown in Table 26. There are several indicators of Freshwater Quality that meet the evaluation criteria. These include measures of contamination, nutrients, and general water condition. Generally, the indicators evaluated under Physical/Chemical parameters performed well under the Primary Considerations, and the Data Consideration with the exception that they were often limited in the spatial and historical extent of the data. No indicators have yet been evaluated under Toxic Organic and Inorganic Chemicals.

Table 26. Summary of Freshwater Quality indicator evaluations. The numerical value under each consideration represents the number of evaluation criteria supported by peer-reviewed literature. For example, the indicator Nutrient Loadings from Rivers to Puget Sound has peer-reviewed literature supporting 2 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary
Freshwater Quality				
Hydrodynamics				
see Water Quantity				
Freshwater Quality				
Physical/Chemical Parameters				
Sediment loadings rate				
Water Quality Index				
Nutrient loadings in rivers to Puget Sound	2	6	3	Nutrient loading to marine photic zone may be significant though possibly less important to overall N when compared to marine sources. Nutrient concentrations in streams is affected by land-use changes, though relationship is complex. Management actions are limited against non-point sources. Effects of nutrient loading sometime complex. Depending on receiving water, change in nutrient loading can affect eutrophication in a predictable manner
Trophic State Index - total phosphorous in lakes				
Dissolved Oxygen	4	5	5	DO levels have clear effects on biota in rivers and streams. DO effected by nutrients. Management actions are limited against non-point nutrient sources.
Water Temperature	4	5	5	Elevated temperatures have clear effects on biota in rivers and streams. Temperature may be controlled by riparian vegetation and/or stream flows. Management options may be complex.
Stream water quality parameters				
Spawning habitat water quality				
Lake water quality parameters - P, N, TSS, chl a,				
Stream C and N flow				
Watershed nutrient hot spots				
Freshwater Quality				
Trace Inorganic and Organic Chemicals				
Toxics in freshwater fish (multiple sources)				
Prespawn Mortality in Coho Salmon				
Toxics in water				
Toxics in freshwater fish (air deposition source)				
Fecal bacteria (streams)				
Indicator	**Primary Considerations (5)**	**Data Considerations (8)**	**Other Considerations (5)**	**Summary**
Biological Water Quality Index				

Indicators for freshwater hydrodynamics were evaluated under Freshwater Quantity – Surface Water Hydrologic Regime.

Next Step: Time constraints prevented a full evaluation of all water quality indicators in marine, freshwater and interface environments. An important next step is to complete the evaluation of water quality indicators.

Water Quantity Evaluation

There are over seventy USGS gauging stations on unregulated rivers and streams in Puget Sound, which are continuously collecting streamflow data. There are over 170 specific metrics that can be used to evaluate different aspects of streamflow. In order to determine which of these is most suitable for Puget Sound, we performed a review of the literature to determine salient management and scientific issues. The management issues of concern and potential indicators are listed below:

Management Issue	**Possible Indicator**
Climate Change	Stream hydrographs, Summer 7-day Annual Low Flow, Center of Timing (CT) of Annual Flow, Spring Snowpack (April 1 Snow-Water Equivalents)
Land use changes/urbanization:	Summer 7-day Annual Low Flow, Peak Flow, Flashiness (High Pulse Count)
Ecology	See above, Violations of Instream Flow Rules

These indicators and others were evaluated as described above. A summary of results is shown Table 27, Table 28, and Table 29. There are many possible indicators of Water Quantity that meet the evaluation criteria.

Table 27. Summary of Freshwater Quantity - Surface Water Hydrologic Regime indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Frequency of flood events has peer-reviewed literature supporting 4 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Surface water hydrologic regime				
High pulse count-	1	7	1	A good measure of flashiness which is a predicted alteration with urbanization/imperviousness. There are demonstrated correlations with Benthic Index of Biological Integrity though not with species of management concern. Management options to reduce hydrologic effects of land use change are limited. Good data.
T_{Qmean} -	1	7	1	A good measure of flashiness which is a predicted alteration with urbanization/imperviousness. There are demonstrated correlations with Benthic Index of Biological Integrity though not with species of management concern. Management options to reduce hydrologic effects of land use change are limited. Good data.
Degree of hydrologic alteration	0			Theoretically unsound. Not a clearly defined single measure, which can be utilized as an indicator.
Annual maximum daily flow / Winter peak flow	3	6	4	Increase in peak flows correlated with land use change and predicted result of climate change. May be important in salmon ecology. Important in flooding. Management options for mitigative actions are limited (particularly with climate change). Good data with the exception that some gauge station perform poorly with high flows. Possibly redundant with Occurrence of Peak Flows and Flooding Frequency.
Number of minimum flow days for each water year	2	8	4	Low flows predicted to increase due to effects of climate change, land use changes, and increased consumptive withdrawals. Important to water resource managers. Good data, though single drought event may disproportionally affect trends. Redundant with 7-day average low flows.
Occurrence of highest flow events per year	2	7	3	Increase in peak flows correlated with land use change and predicted result of climate change. May be important in salmon ecology. Peak flows more descriptive of flooding and flow timing. Good at demonstrating long term trends. Possibly redundant with Annual Maximum Flow and Flooding Frequency.
Spawning flows	1	8	1	Flows during spawning period may affect water temperature, habitat availabiliy, and energetics. Conditions vary depending on salmon run and river. Clear flow-response relationships not established due to potentially conflicting factors. Spawning flows may need to be definied for individual reaches and/or individual salmon runs. Salmon health of high management concern. Good data.
Percent of flows that create & maintain habitat	0			Theoretically unsound. Establishing flow-habitat relationships are complex and difficult to define. May vary between streams and reaches. Typically done for single species. Different species/habitat may require different aspects of flow for establishment (e.g. riparian vegetation require peak flows). Change in indicator may not be descriptive of important changes.
Percent of flows that meet summer base flows to support species	0			Theoretically unsound. Difficult to define due to the myriad of important habitats and the unique flow/habitat relationships that may exist on each river.
Annual mean flow streams and rivers	3	8	4	Important to water resource managers. May be affected by increased consumptive use. Limited magement options mainly concerning conservation and reuse. Good data. Indicator more descriptive when combined with other indicators of hydrologic alteration.
April and May Snow Water Equivalents (SWE), Spring Snowpack	3	6	4	Observed past and predicted future decreasing trends due to climate change. Important to water resource managers. Long term changes would alter flow regimes, which is potentially ecologically important. Management resoponses limited. Good data. Can be complimentary or redundant (7-day low flow, flow timing) depending on suite of indicators.
Glacier mass balance	2	4	3	Observed and predicted future changes due to climate change. Important to water resource manager. Long term changes would alter flow regimes, which is potentially ecologically important. Management resoponses limited. Moderate data.
Annual Center of Timing (CT)	3	7	2	Observed and predicted future changes due to climate change. Important to water resource manager. Long term changes would alter flow regimes, which is potentially ecologically important. Management resoponses very limited. Good data. Good complimentary with other indicators of hydrologic alteration.
Violations of DOE instream flows	3	8	3	Good indicator of management effectiveness. Instream flow rules may not be proctective of ecology. Good range of possible management responses. Good flow data. Instream flow rule only established on limited number of streams in Puget Sound. Somewhat redundant with 7-day Average Low Flow and Number of Minimum Day Flows per Year
Storm water quantity	Not yet evaluated			
Frequency of flood events	4	7	4	Predicted increased flooding with urbanization due to higher runoff from impervious surfaces. Higher winter flooding due to climate change due to more winter rain instead of snow, and rain-on-snow events. Important to management. Limited management responses. Established floodstage targets. Good flow data. Possibly redundant with Annual Maximum Flows or Occurency of High Flow Events.

Table 28. Summary of Freshwater Quantity – Groundwater Levels and Flow indicator evaluations. The numerical value that appears under each of the considerations represents the number of evaluation criteria supported by peer-reviewed literature. For example, Annual 7-day low flow has peer-reviewed literature supporting 3 out of 5 Primary Considerations criteria. Details can be found in the accompanying spreadsheets.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Groundwater levels and flow				
Groundwater elevation/flows	Not yet evaluated			
Annual 7-day low flow	3	8	5	Predicted decrease in summer flows with climate change and increased consumptive use. Several studies show GW/surface water interactions with potential implications of low flows. Important ecologically. Important to management. Limited management responses. Good data. Complimentary with other indicators of the hydrologic flow regime. Somewhat redundant with Violations of Instream Flow and Number of Minimum Day Flows per Year.

Table 29. Summary of Freshwater Quantity – Groundwater Levels and Flow indicator evaluations.

Indicator	Primary Considerations (5)	Data Considerations (8)	Other Considerations (5)	Summary Comments
Consumptive water use and supply				
Storage days remaining	Not yet evaluated			
Water use/demand	Not yet evaluated			
Summer/autumn reservoir inflows	Not yet evaluated			

Surface Water Hydrologic Regime – Overview

The Puget Sound basin includes at least thirteen major river systems and numerous tributaries, which can be classified as rainfall-dominated, snowmelt-dominated, or transitional [191-193]. Rainfall-dominated rivers exhibit peak flows during winter; snowmelt-dominated rivers have peak flows in late-spring and late-fall with low winter flows. Transitional rivers exhibit less pronounced high or low flows in the late-Fall and late-spring, and winter. Hydrologic flow patterns are important both ecologically and in terms of consumptive resources. Alteration of historic flow patterns may cause ecological harm and supply disruptions [23, 80]. Hydrologic flow regimes in Puget Sound rivers have been altered through the construction of dams for flood control or power generation, or by changes in land cover and climate. Flows in the Skagit, Nisqually, Green, Skokomish, and Cedar rivers are regulated by dams [76].

There are over seventy USGS gauging stations on unregulated rivers and streams in Puget Sound. As such, there are ample data available for flow analysis and it is possible to use this data to evaluate streamflow patterns in many different ways. In order to determine which is the best way to analyze the data it is important to consider what are the most significant ecological and management concerns of the region. The bulk of this section presents a literature review that is intended to determine the important management and ecological issues of Puget Sound.

Indicators of Hydrologic Alteration

The surface water hydrologic regime of a river or stream can be characterized through measures of magnitude, frequency, duration, timing, and rate of change [174]. At least 170 specific metrics have been used to describe specific aspects of the hydrologic regime resulting in the potential for considerable redundancy [108]. The most suitable metric, or suite of metrics, is dependant on the specific nature of the question being addressed or the issues that are of greatest management concern [32, 63, 64].

The Puget Sound Partnership (PSP) has identified the following issues of potential concern related to water quantity in Puget Sound:

- Consumptive use of surface and groundwater;
- Changes in hydrology related to land use;
- Climate change;
- Modification to stream and floodplain habitats [125]

A stated goal of the management of water quantity in Puget Sound is:

- In-stream flows directly support individual species and food webs, and the habitats on which they depend [1].

The intent of this section is to describe the process of determining an appropriate set of indicators of hydrologic alteration, which are relevant to management concerns. Indicators will also be screened according to the criteria discussed elsewhere in this Puget Sound Science Update.

The following sections describe a review of the recent literature with geographic focus on Puget Sound. There were two objectives of the literature review: 1) determine which of the indicators of hydraulic alteration would be most appropriate based on the predicted or observed alternations related to land use change and climate change, and 2) determine which aspects of the flow regime are known to be most relevant to the aquatic species in Puget Sound streams and rivers.

Discussions of consumptive water use and habitat alterations are elsewhere.

Indicators of Hydrologic Alteration – Climate Change

Indicators of Hydrologic Alteration – Climate Change – Summary

- Analysis of historic streamflow data in the Western United States suggest that spring snowpack is decreasing and streamflow timing is getting earlier in the water year. These trends are apparent despite significant annual and systematic variation associated with the El Niño/Southern Oscillation and the Pacific Decadal Oscillation.

- Temperatures in the Puget Sound region are projected to increase an average of approximately 0.3°C per decade over the 21st century due to climate change.
- Increasing temperatures may lead to decreased spring snowpack, earlier spring runoff, and lower summer flows.
- Climate change associated hydrologic alterations may lead from snowmelt or transition (snow-rain) flow patterns to rainfall dominated flow patterns.
- Decline in snowpack may be problematic for regional water supplies as most systems have been developed base on historic flow patterns [194]

Indicators of Hydrologic Alteration – Climate Change – Literature Review

Puget Sound river hydrology may be affected by climate change. Precipitation in the region occurs predominately in the winter months. The accumulation of snow in the mountains is a primary storage mechanism particularly for the snowmelt-dominated and transitional river systems. It has been estimated that upwards of 70% of total stream discharge in the Western United States is from melting snowpack [192]. An estimated 27% of summer streamflow of the Nooksack river originates from high-elevation snowshed and glacier-derived meltwater [193]. Climate change assessments have predicted increased winter and spring temperatures resulting in decreased snowpack storage in the mountains, increased winter runoff as more precipitation falls as rain, and lower summer flows [83, 192, 197-200]. Climate change may force rivers with snowmelt-dominated and transitional hydrological flow patterns toward rainfall-dominated hydrology [194].

Prediction of the regional impacts of climate change on river and stream hydrology can be confounded by typical variation in rainfall patterns, high geographic variability, and land use changes. There are at least two large-scale systems that affect the annual climate variations in the Pacific Northwest [201]. The El Niño/Southern Oscillation, with a period of 2 to 7 years, and the Pacific Decadal Oscillation (PDO), with an estimated half-period of 20 to 30 years. Warm and cool phases of the El Niño/Southern Oscillation and/or Pacific Decadal Oscillation may result in variations on the order of 1°C for temperature, and 20% for precipitation [201]. Hamlet et al. (2005) utilized a Variable Infiltration Capacity model to discern long-term trends in spring snowpack from temperature and precipitation variability [195]. They found that downward trends in snowpack associated with temperature were related to widespread warming. Trends of snowpack associated with precipitation were largely controlled by decadal oscillations; climate change effects on precipitation have not been detected [196].

Mote et al. (2008) concluded that the primary factor in decreasing snowpack in the Washington Cascades was rising temperatures, consistent with the global warming [196]. The long-term snowpack trends were unrelated to the variability brought about by Pacific oscillations (e.g., PDO).

Casola et al. (2009) investigated the potential impacts of climate change on snowpack by combining future temperature predictions with the estimated temperature sensitivity of spring snowpack [203]. They utilized four distinct methods to estimate sensitivity and all four converged on a result of approximately 20% loss in spring snowpack per 1°C temperature rise. Analysis of historic and projected temperature data indicated that snowpack reductions over the past 30 years ranged from 8%-16% while future temperature change would result in an 11%-21% reduction in spring snowpack by 2050. However, future trends may not be statistically detectable due to a high level of interannual variability.

Barnett et al. (2008) utilized a multivariate analysis to evaluate the simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999 [83]. They found significant increasing trends in winter temperature, and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt). In order to distinguish natural variation from anthropogenic forcing they evaluated the observations against two separate climate models and found that the hydrologic changes were both detectable and attributable to anthropogenic forcing.

Stewart et al. (2004) investigated historic (1948-2000) and future streamflow timing in snowmelt dominated rivers and streams in the Western United States [197]. They found significant trends towards earlier runoff in many rivers and streams in the Pacific Northwest. Utilizing a 'Business-as-Usual' emissions scenario with a Parallel Climate Model, they predicted a continuation of this trend, largely due to increased winter and spring temperatures but not changes in precipitation. In a companion study they further analyzed the trends in streamflow timing with variations of the PDO [198]. While streamflow timing was partially controlled by the PDO there remained a significant part of the variation in timing that was explained by a longer-term warming trend in spring temperatures.

Luce and Holden (2009) utilized quartile regression to investigate the trends in streamflow in wet (75th percentile), dry (25th percentile), and average (50th percentile) water years in rivers in the Pacific Northwest [199]. They reported that the highest proportion of significant decreasing trends occurred during the dry years, while there were few significant trends in the high flow years, concluding that the dry years were getting dryer in the Pacific Northwest. This aspect of the trends accounted for much of the increased variability in annual streamflow.

Recently, the Climate Impact Group, part of the Joint Institute for the Study of the Atmosphere and Ocean (JISAO) at the University of Washington performed The Washington Climate Change Impact Assessment. The assessment included analyses on hydrology and water resource management in which they utilized results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff [200]. For the rivers in the Puget Sound basin they found a dramatic decrease in spring snowpack with there being almost no April 1 snowpack by 2080. During that period, river hydrographs progressively changed from transition or snow-rain dominated to rain dominated patterns. There was little predicted change in annual precipitation.

Indicators of Hydrologic Alteration – Climate Change - Relevant Indicators

Based on the review of the literature, the following indicators of hydrologic alteration may be suitable to monitor and evaluate potential changes in the hydrologic regime brought about by climate change:

- Stream hydrographs
- Summer 7-day Annual Low Flow
- Center of Timing (CT) of Annual Flow
- Spring Snowpack (April 1 Snow-Water Equivalents)

Indicators of Hydrologic Alteration – Land Use/Urbanization

Indicators of Hydrologic Alteration – Land Use/Urbanization – Summary

- Puget Sound region has experienced extensive development and urbanization. The population of the 12 counties surrounding Puget Sound was approximately 4.2 million in 2005; it is expected to increase to 5.5 million by 2025 [201].
- Land use changes associated with increases in population affect river and stream hydrology. Typical changes include reduced infiltration and increased runoff, increased flashiness, and decrease in summer flows.

Indicators of Hydrologic Alteration – Land Use/Urbanization – Literature review

Alterations in land use can affect stream and river hydrology in various ways (see [80] and references therein). Urbanization is associated with the increase of impervious surface area, which can result in increases the severity and frequency of peak stream flows by reducing infiltration and increasing runoff; overall annual stream flow volumes are generally not affected [209-215]. Urbanization my lead to lower base flows from reduced infiltration, though this effect can be somewhat offset by a reduction in evapotranspiration from the clearing of trees [212]. The construction of storm drain systems has been implicated as a primary factor in the reduction a base flows [202]. Logging of forested lands increases annual flow by reducing evapotranspiration in the watershed though other hydrologic changes such increasing flooding are disputed [217-219]. River basin land use alterations may lead to alterations in channel morphology which can exacerbate flooding potential without changes in stream flow [203].

Burges et al. (1998) compared hydrology from a forested and a developed basin in Puget Sound lowlands [204]. They found that surface runoff accounted for 12%-30% and 44%-48% of rainfall on forested and developed catchments, respectively, suggesting that the rate of infiltration was much higher in the forested basin. In a similar study, Leith and Whitfield (2000) found an increased streamflow in basins with the most increase in urbanization compared to basins with less development [205]. Moscript and Montgomery (1997) found an increased flood frequency in streams with urbanized watersheds compared to nearby control watersheds, which had not undergone development [206].

Konrad and Booth (2002) investigated possible hydrologic effects related to urbanization by evaluating stream flow statistics from ten streams in the Puget Sound basin [207]. They found that the fraction of the year that flow was above average annual flow (TQ,mean) and the maximum annual flow (Qmax) had significant trends in the urbanized basins compared to the rural basins and could be useful in monitoring the effects of urbanization on stream hydrology. They suggested that TQ,mean might be of more practical use. Fleming (2007) analyzed the effects of urbanization by examining stream memory (i.e. the effect of prior stream flow on current discharge) in urbanizing and rural watersheds in the Puget Sound lowlands [208]. He reported that memory decreased in the developed basin over time but not the undeveloped basins, suggesting that flow memory would be a useful measure of development in a watershed, though may be dependent on basin size, with larger basins exhibiting a greater fidelity in memory.

Cuo et al. (2009) utilized a Distributed Hydrology-Soil-Vegetation Model in order to determine the relative effects of land cover and temperature change on the flow patterns in Puget Sound streams [211]. They found that the relative importance of temperature and land cover differed between the upland and lowland basins. In the lowland basins land cover changes were more important and generally resulted in higher peak flows and lower summer flows primarily from increased runoff. Both land use change and climate effects were more important in the upland basins. Climate effects were more important in the transitional zones and resulted in higher winter flows, earlier spring peak flows, and lower summer flows.

Indicators of Hydrologic Alteration – Land Use/Urbanizations - Relevant Indicators

Based on the review of the literature, the following indicators of hydrologic alteration may be suitable to monitor and evaluate potential changes in the hydrologic regime brought about by land use/urbanization:

- Summer 7-day Annual Low Flow
- Peak Flow
- Flashiness (High Pulse Count)

Hydrologic Regime – Ecology 5.5.2.2.5 Hydrologic Regime – Ecology – Summary

- Aquatic species in Puget Sound rivers and streams are generally adapted to historic flow patterns.
- Salmonid species appear to be sensitive to land use changes in watersheds with streams in urban areas being associated with less robust populations of coho compared to forested areas.
- Benthic invertebrate communities appear to be negatively affected by increased flashiness of stream hydrology associated with urbanization.

Hydrologic Regime – Ecology – Literature Review

The alterations of river and stream hydrology can affect aquatic ecosystems by changing physical habitats, disrupting the natural connectivity of habitats, or by facilitating the successful invasion of exotic species [224]. Native species may have evolved according to the pressures and timing of natural flow regimes; altering flow patterns may negatively affect those species [225]. However, it is not always possible to separate the biological impacts of altered river or stream hydrology from the biological impacts associated with the land-use changes that often accompany or force the alteration in hydrology.

Several studies have attempted to evaluate the ecological impacts of altered land use in stream and river watersheds in Puget Sound. Spawner survey data collected by Moscript and Montgomery (1997) suggested a decline in salmon populations in basins that underwent urbanization, but not in a nearby control basin [206]. Scott et al. (1986) compared fish populations in a urbanized stream with a nearby unaffected control stream and found that while overall fish biomass was similar between the two sample sites there were differences in species composition [209]. The urbanized stream population was dominated by cutthroat trout while the control stream population consisted of a wide array of salmonids, including coho, and non-salmonids.

Pess et al. (2002) performed a broad-scale analysis over 16 years to investigate salmon abundance with land use and habitat in the Snohomish river basin [210]. The proportion of adult coho supported by a particular stream reach was consistent over the course of the study and the median adult coho density was consistently higher in the forested areas compared to the more-developed areas.

Bilby and Mollot (2008) compared the distribution of spawning coho salmon in four Puget Sound rivers with changes in land use between 1984 and 1991 [211]. They found that, while the overall numbers of spawning coho changed at all sites, there was an approximately 75% reduction in the proportion of salmon spawning in areas of increased urban land use as well as a smaller decline in areas with increased agricultural land use activities. They suggested that the protection of spawning habitat may be important.

While these studies demonstrate relationships between urbanization and ecology, and urbanization has been shown to affect stream hydrology, there are several other factors, including an increase in contamination input from surface runoff and habitat modification, which likely influence the results [212]. There are several are several other studies which have attempted to elucidate the specific effects of hydrologic changes on in-stream ecology, including fish and benthic invertebrates; these are discussed below.

High flows can affect salmon returns by disrupting redds, increasing deposition of fine sediments and reducing dissolved oxygen transfer, reducing growth rates, or increasing downstream displacement and mortality [225]. In a Puget Sound stream, egg burial depths were observed to be slightly deeper than typical scour depths caused by flooding during the incubation period suggesting an adaptation to environmental flow conditions [213]. Increases in peak flow due to land development or other causes may then significantly contribute to embryo mortality. Schuett-Hames et al. (2000) also investigated scour depth in two locations in a Puget Sound lowland stream [214]. They observed sediment scour during two storm events with estimated return intervals of 1 and 1.4 years and found that scour depths reached median egg pocket depths at 20% of the monitored sites during the larger storm. This suggests that scour related to high flows may be important in salmon mortality in Puget Sound.

Beamish et al. (1994) identified an inverse relation between anomalously high flows and indicies of production for coho and Chinook salmon in the Frasier River but not for chum, pink, or sockeye salmon suggesting that, at least in some cases, extreme flows may affect survival [215]. They did not identify a causative mechanism.

Greene et al. (2005) utilized standard multiple regression analysis to evaluate correlations between various environmental factors in the freshwater, bay/delta, and ocean habitats and the return rates of Chinook salmon in the Skagit River [232]. Their results indicated that flood magnitude, as measured though the Flood Recurrence Interval of the peak flow during incubation period, was a strong predictor of the return rate for Chinook salmon; there was a negative correlation between flood magnitude and salmon returns. A bay habitat factor, which was calculated based on measures of sea level, sea level pressure, and upwelling, was also significantly correlated with Chinook return rates.

In order to evaluate the overall effects of anthropogenic changes on salmon abundance, Scheuerell et al. (2006) utilized a multistage model to incorporate population growth, habitat attributes, hatchery operations, and harvest management based on predictive relationships from the published literature [233]. Relationships between peak daily flow during incubation period to egg-to-fry survival rate for Chinook or sockeye have been reported for Puget Sound rivers [234-237]. Although the reported data generally indicate a decrease in egg-to-fry survival with increasing peak flow during incubation period, the apparent best-fit regression (i.e. negative exponential, logarithmic, or linear) varies, demonstrating the uncertainty in the relationship. Battin et al. (2007) utilized the same relationship but also considered the potential limitations on spawning capacity that could be brought about by minimum flows during the spawning period [216]. They found that the model results were relatively insensitive to spawning capacity (and minimum flows).

Summer flows have been shown to be correlated with coho run strength in Puget Sound [217].

Bauer and Ralph (2001) evaluated the potential utility of incorporating aquatic habitat indicators, including those related to flow regime, into legal standards for water quality [218]. However, they concluded that the effects of low flow on habitat availability was sufficiently well understood to only allow the development of narrative, but not numeric criteria; the relationships between peak flows and habitat were less certain.

Similarly, Poff et al.(2010) recently reviewed 165 papers to investigate the possibility of developing quantitative relationships between various types of hydrologic alteration and ecological response [81]. While there was a general reported decline in ecological metrics in response to changes in flow metrics, including a general decline in fish abundance and diversity with alterations in flow magnitude, they were unable to support any quantitative relationships.

Matzen and Berge (2008) evaluated the relationship between urbanization and fish populations in Puget Sound lowland streams through the development of a fish index of biotic integrity (F-IBI; [219]). Due to the low species diversity characteristic of Puget Sound lowland streams, they utilized several metrics, which were specific to the region; the final F-IBI included a combination six metrics, which showed the strongest correlation to TIA. The authors cautioned against the direct comparison of individual IBI scores, or the value of short-term trends due to the likelihood of spatial or temporal variation that can occur within streams.

There are several studies that evaluate the effects of urbanization on stream condition based on a benthic index of biological integrity (B-IBI). Morley and Karr (2002) investigated the relationships between stream biological condition, as measured by the B-IBI, and the extent and distribution of urbanization, and stream flow in Puget Sound lowland streams [220]. They reported that B-IBI was significantly correlated with urbanization, as measured by percent urban area and percent impervious area in a sub-basin. Further, they found that B-IBI was correlated with measures of flashiness though not peak flow, and relative roughness though not measures of pebble or fine diameter (e.g. D16 or D50). Based on these relationships they argued that benthic invertebrates were a key measure of stream condition, though not necessarily predictive of the condition of fish populations.

Booth et al. (2004) reported similar correlations between B-IBI and percent urbanization, percent imperviousness, and several measures of flashiness [213]. They did not conclude that urbanization would be a good predictor of stream health but rather suggested that levels of urbanization may constrain the potential benthic diversity of a particular stream and that urbanization may affect each stream differently.

Bond and Downes (2003) performed a set of controlled studies and found that flow increases, but not changes in fine sediment transport, were sufficient to disturb benthic communities in streams, though the effects may be dependent on the availability of flow refugia [221]. This is consistent with studies, which suggest that benthic diversity is sensitive to hydrologic alterations brought about by urbanization.

King County investigated the relationships between flow alterations and in-stream ecology in Puget Sound lowland streams through the Normative Flow Project [222]. They used data from a set of locations representing a range of land cover conditions to evaluate the effects of land use on hydrology and biological condition, as measured through the B-IBI and other macroinvertebrate metrics. The hydrologic metrics with the strongest correlation with B-IBI included low-flow threshold pulse events and interval between pulses, high-flow threshold pulse events and total period of the year with high pulses, TQmean, percent of time above the mean two-year flow, and timing of the onset of fall flows. Although none of the hydrologic indicators were good predictor of B-IBI they were able to discriminate the difference between high and low B-IBI values.

Alberti et al. (2007) evaluated the patterns and connectivity of urbanization by performing an empirical analysis of land use intensity, land cover composition, landscape configuration, and connectivity of the impervious area, on B-IBI in Puget Sound lowland streams [245]. Their analysis suggested that total impervious area (TIA) explained much of the variance in B-IBI across basins, but other factors such as mean patch size of urban land cover and number of roads crossing a stream could explain part of the variance not explained by TIA alone. They also reported an inverse relationship between the aggregation of forested land and B-IBI suggesting that intact forests are important to benthic diversity.

DeGasperi et al. (2009) performed a retrospective analysis to relate measures of hydrologic alteration that were sensitive with measures of urbanization and benthic diversity, but not sensitive to basin area [106]. They found that high pulse count (the discrete number of high pulses per water year when flow is exceeds twice the average annual flow rate) and high pulse range (the number of days from the first high pulse to the last high pulse in the water year) best fit their evaluation criteria. Their analysis suggested as a basin is urbanized the number of high pulses increase in the winter and are more likely to occur in the summer increasing both the discrete number of pulses and the range. These pulses affect appear to affect B-IBI values.

Although the B-IBI score may be correlated with specific types of hydraulic alteration which specifically affect benthic communities, there is no clear relationship between B-IBI and the condition of vertebrate species [220]. Further, the natural variability of biological indices has not been well characterized; large variability may lead to inaccurate determinations of river health [246]. There can be both large and small scale spatial variability as well seasonal and inter-annual variability, all of which needs to be well understood in order to correctly attribute changes in biological condition with physical alteration brought about by anthropogenic activities. Mazor et al. (2009) found fluctuating conditions at sights without obvious changing conditions suggesting that short-term bioassessments may lead to inaccurate conclusions [246].

Summary of Water Quantity Indicators

A summary of the indicator evaluation in presented in Table 27, Table 28, and Table 29. In summary there is a wide range of possible indicators of the Surface Water Hydrologic Regime, which perform very well under both the Primary and Data Considerations. There is ample data for the region that can be parsed and evaluated in many different ways. It is, therefore, essential to understand the management concern or objective prior to indicator selection to ensure that the indicator is appropriate to the question at hand.

Only a single indicator was evaluated for groundwater levels and flows. It performed well against the Primary and Data considerations. However, owing to subsurface heterogeneity, the spatial variation is often not well understood, nor is it possible to confidently infer condition at on location from based on data collected proximally.

No indicators were completely evaluated for consumptive use and supply. However, a preliminary review suggests that there are good performing indicators, though it may be a time-consuming task to collect and compile the data on a regional scale.

Key Point: There is ample data to support the use and continued development of water quantity indicators. However, different indicators will better form different management concerns or objectives. Thus, prior to indicator selection it is critical to precisely define the management goal and operational objectives.

Ranking Puget Sound Indicators

Terminology and concepts	
Ecosystem assessment indicator	Technically robust and rigorous metric used by scientists and managers to understand of ecosystem structure and function
Improving indicator	Indicator that is increasing faster in the short-term but slower in the long-term than an index that captures aggregate changes in multiple indicators
Lagging indicator	Indicator that is increasing slower in the short- and long-term than an index that captures aggregate changes in multiple indicators
Leading indicator	Indicator that is increasing faster in the short- and long-term than an index that captures aggregate changes in multiple indicators
Other considerations	Indicator evaluation criteria that make an indicator useful, but without which an indicator remains scientifically informative
Ranking scheme	Approach used to weight indicator evaluation criteria
Slipping indicator	Indicator that is increasing faster in the long-term but slower in the short-term than an index that captures aggregate changes in multiple indicators
Vital sign indicator	Scientifically meaningful, but simple, metric that can generally inform the public and policy makers about the state of the ecosystem

The matrix of ecosystem indicators and indicator evaluation criteria provides the basis for ranking indicators. However, ranking indicators requires careful consideration of the relative importance of evaluation criteria. The importance of the criteria will certainly vary depending on the context within which the indicators are used and the people using them. Thus, ranking requires that managers and scientists work together to weight criteria. Failure to weight criteria is, of course, a decision to weight all criteria equally.

As an example of how our matrix could be used to rank indicators, we compare two food web indicators, ratfish/flatfish and jellyfish, using different weighting schemes. We provide these examples simply as an illustration, not to advocate one weighting scheme versus another.

One could begin by scoring each indicator as 1.0 when there is peer-reviewed evidence that that it met a criterion. When there is non-peer reviewed or ambiguous evidence that an indicator meets a criterion we give it a score of 0.5. When it does not meet a criterion, it receives a score of 0.

Equal weights: In this first scheme, we weight all criteria equally. In this case, ratfish/flatfish get a score of 10.5, while jellyfish score a 10 (out of a possible 19).

New monitoring programs: Imagine, however, a case in which the availability of historical data is less important (e.g., when considering a new monitoring program). In this instance, one might wish to ignore data considerations such as “historical data available”, “broad spatial coverage”, “continuous time series”, and “variation understood”. In this scheme, the ranking of the indicators reverses with jellyfish scoring 9.5, while ratfish/flatfish score 8.5 (out of 15).

Discounting importance of peer-review: Our initial weighting discounts indicators that were not supported by peer-reviewed evidence. It is conceivable that in some settings practitioners might wish to equally weight non-peer and peer reviewed evidence. In this case, because much of the evidence supporting the data criteria for ratfish/flatfish is not in peer-reviewed literature, the score for this indicator would increase to 14.5 (out of 19).

Whatever ranking scheme is used, our matrix can serve as a useful starting place for sorting through large numbers of indicators. By carefully ranking indicators in a manner consistent with specific management and policy needs, and choosing to focus on high-ranked indicators for each attribute, a winnowing of indicators naturally takes place.

Specificity and sensitivity of indicators

Long lists of indicators can present challenges for drawing inference about overall ecosystem status. A useful way to interpret lists of indicators in aggregate focuses on one of the primary considerations in the set of evaluation criteria introduced above, “the indicator responds predictably and is sufficiently sensitive to a specific ecosystem attribute.” Two of the terms in this criterion, “specific” and “sensitive,” can be used to organize indicators according to the type of information they provide about attributes. Rapport et al. (1985) proposed that an indicator’s specificity can be distinguished based on whether it reliably tracks few or many attributes [5]. An indicator that provides information about many attributes (even attributes of multiple PSP goals) is non-specific but perhaps broadly informative of ecosystem status. An indicator that serves well as a proxy for fewer attributes can be thought of as diagnostic of changes in specific ecosystem characteristics. For example, in Figure 8 harbor seals are a non-specific indicator for Species and Food Webs attributes whereas jellyfish are a diagnostic one.

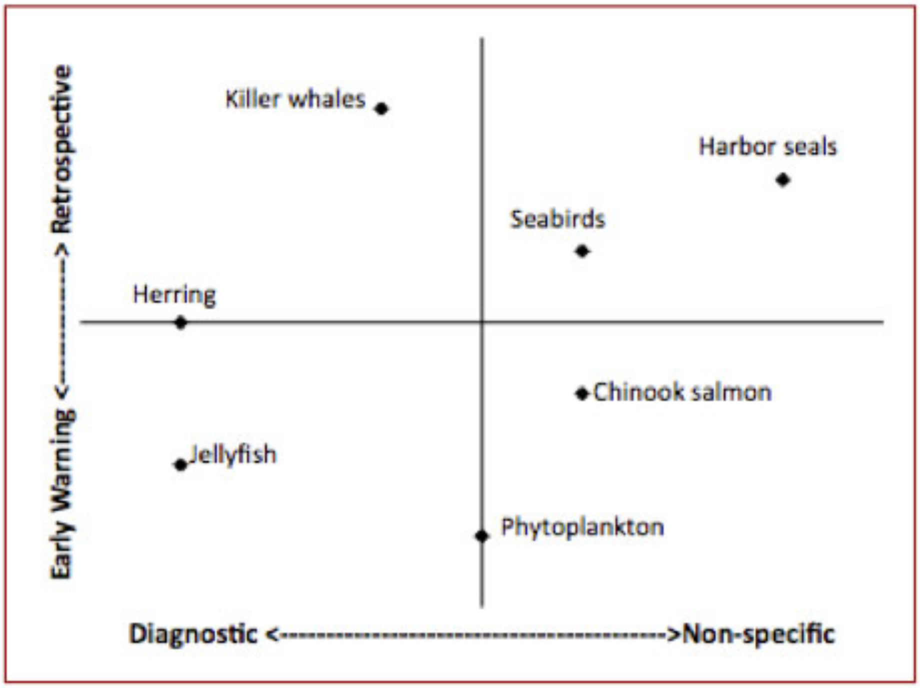
Another informative axis on which to interpret an indicator is in terms of its sensitivity. An indicator that provides information about impending changes in attributes before they occur is an early warning or “leading” indicator. For instance, due to fast turnover rates, phytoplankton are likely to be an early warning indicator for Species and Food Web attributes in Puget Sound (Figure 8). In contrast, an indicator that reflects changes in attributes only after they have occurred is a retrospective or “lagging” indicator. Retrospective indicators, such as killer whales (Figure 8), are likely to be characterized by slow turnover rates, but can nonetheless be useful for interpreting cumulative impacts and ecosystem-wide shifts in attribute values.

Vital Sign vs. Assessment Indicators

Ranking schemes provide a mechanism for narrowing the long list of indicators presented above to a more manageable set that facilitates inference about the status of the Puget Sound ecosystem. Here we suggest that focusing on the specificity and sensitivity of an indicator, in combination with its performance against the “understood by the public and policymakers” criterion introduced above, provides a framework for reporting on the status of Puget Sound.

Previous indicator development efforts in the Puget Sound region (e.g. [34]) and beyond (e.g., [223]) have advocated a two-pronged approach to indicator reporting. Recchia and Whiteman (2009) refer to a coarse-grained evaluation of ecosystem status and trends. This level of indicator reporting is aimed at the general public and policy makers with the goal of providing a limited number of “vital signs” of the ecosystem [223]. Vital Signs may not be very specific, and they do not need to be sensitive on any particular time scale. For instance, abnormalities in blood pressure or temperature indicate some malady, but do not suggest a specific pathology. Likewise, changes in Chinook salmon abundance may be brought about by alterations to water quality, habitat, climatic factors, fishing or numerous other factors, in the marine, freshwater, or terrestrial domains of Puget Sound. Nonetheless, it is likely that changes in Chinook salmon represent a shift in the “health” of the system (Figure 8). As regional managers and scientists consider assembling portfolios of Vital Sign indicators, some indicator criteria may be more important than others. For example, it is clearly crucial that the indicator be understandable to the general public. On the other hand, understanding the variance structure of such indicators may be less critical. By carefully crafting a weighting scheme as described in Section 5.6, it is possible to systematically sift through a large inventory of indicators to generate a short-list of scientifically credible vital sign indicators. Ultimately, the goal of Vital Sign indicators is to provide a limited number of scientifically meaningful, but simple metrics that can generally inform the public and policy makers about the state of the ecosystem.

Figure 8. Indicator species in Puget Sound plotted according to whether they reliably track few (diagnostic) or many (non-specific) Species and Food Web attributes (x-axis) and whether they respond quickly (early warning) or slowly (retrospective) to perturbations. The ranking of indicators as diagnostic vs. non-specific is relative and based on the analysis in [118]. The ranking of indicators as early warning vs. retrospective is also relative, and based on the production to biomass ratios of these seven species. Adapted from [5].



In contrast to Vital Sign indicators, Ecosystem Assessment indicators provide a technically more robust and rigorous understanding of ecosystem structure and function. Assessment indicators provide the detailed information necessary to diagnose specific problems, develop strategies to mitigate these problems, and monitor responses of the ecosystem to management actions on multiple time scales. Thus, Ecosystem Assessment indicators should be diagnostic rather than non-specific, but can span a range of sensitivities, so that a full set includes both early warning and retrospective indicators. The audience for these indicators is scientists and managers who require a detailed understanding of the ecosystem; consequently, criteria related to the technical performance of the indicator should be given increased weight relative to criteria related to salience.

Key Point: Ranking indicators requires careful consideration of the relative importance of evaluation criteria. The importance of the criteria will certainly vary depending on the context within which the indicators are used and the people using them. Thus, ranking requires that managers and scientists work together to weight criteria. Weighting schemes that emphasize communication will inform the selection of Vital Sign indicators, while weightings that stress technical aspects of the data will inform the selection of Ecosystem Assessment Indicators.

Defining Ecosystem Reference Levels: A Case in Puget Sound

1. Ecosystem reference levels: how do we know when EBM has succeeded?

Ocean stewardship is not simple. Rather than maintaining piecemeal efforts, scientists, managers, conservationists, and policymakers have agreed that restoration and protection of the oceans will require a more integrated approach [249-251]. A unified appeal for marine ecosystem-based management (EBM) has made the task of developing concrete methods for implementation quite urgent [20, 252, 253]. Indeed, if the goal is maintenance and sustainable use of a healthy ecosystem [224], it follows that those responsible for achieving this objective require a means to track the progress of their efforts. As discussed above, indicators allow the tracking of progress and change.

Terminology and concepts	
Baseline	Reference level derived from time periods or locations free from human pressures
Benchmark	Indicator value suggestive of progress toward targets
Limit	Reference level pegged to an extreme value beyond which undesired change occurs
Nonlinearity	Sudden change in a response variable resulting from smooth and gradual change in a causal factor
Normative reference level	Reference level defined based on what is socially acceptable, i.e., according to norms
Norms	Define what is generally accepted within a cultural context, and may serve as societal standards to evaluate ecosystem conditions, human activities, or management strategies
Reference direction	Which specifies how the trend in an indicator relates to the desired state of the ecosystem
Reference level	Point value or direction of change used to provide context so that changes in indicator values can be interpreted relative to desired ecosystem states
Reference point	Precise values of indicators used to provide context for the current status of an indicator
Target	Reference level that signals a desired state

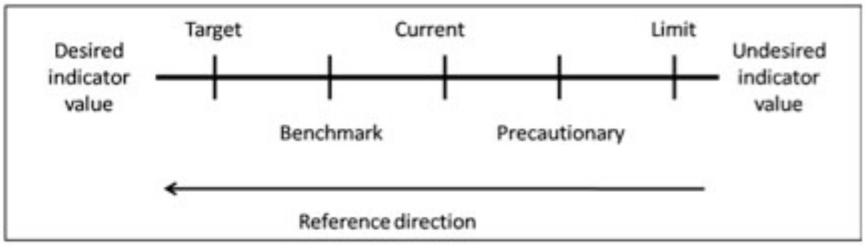
Many authors have considered ecosystem health to be the structure and function of the ecosystem desired by stakeholders in a specific management context [255-258]. Thus, as we have previously emphasized, many attributes of ecosystem health, such as resilience, are difficult to measure directly. Proponents of using human health as an analog to ecosystem health note that just as cholesterol, stress, and income levels can serve as indicators for gauging human health (a state of physical, mental, and social well-being; [225]), the status of an ecosystem's health can be measured via proxy using a suite of ecosystem indicators. For example, it is widely appreciated that the abundances of certain species of jellyfish and top predators provide information about the status of marine ecosystems because they reflect underlying changes in important ecosystem functions (e.g., [226, 227]).

To be useful from a policy and management perspective, ecosystem indicators must be linked to reference levels. Reference levels provide context so that changes in indicator values can be interpreted relative to desired ecosystem states [113, 255, 257, 262]. Following with the human health analogy, one reference level for household income, a social well-being indicator, might be the poverty line [228]. In single-species and single-sector management, reference levels are also fairly well established. Examples include target population sizes for recovery of endangered species [229], the harvest rate corresponding to maximum sustainable yield in a fishery [230], the critical level of nutrient input beyond which a clear freshwater lake becomes turbid [231], and, acceptable concentrations of toxic contaminants in water bodies [232]. While existing reference levels such as these provide a useful starting point [233], EBM requires the consideration of how interactions among species and management sectors affect overall ecosystem state and potential trade-offs among indicator values [234]. Reference levels set to guide management of species, habitats, and water quality individually may need to be modified or supplemented with additional indicators, and corresponding reference levels, in order to steward multiple ecosystem components simultaneously. We believe that many of these challenges can be met by adopting successful approaches from other management contexts for use on the ecosystem level. Here we describe several approaches for linking indicator values and trends to reference levels related to ecosystem health, and provide some examples for how they might be applied in Puget Sound. A summary of existing targets and/or reference levels for Puget Sound follows.

Reference points and reference directions

Reference points are precise values of indicators used to provide context for the current status of an indicator. Establishing a reference point requires substantial understanding of an indicator's properties, but it provides a rigorous way to assess ecosystem status. For some indicators, reference points will have already existed prior to the introduction of EBM. In the case of Puget Sound, the Washington Department of Health provides recommendations regarding human consumption of seafood subject to known levels of toxic contamination [235]. In the short-term, it may be challenging to develop actual point values for ecosystem reference levels [255, 262, 271]. However, a reference direction, which specifies how the trend in an indicator relates to the desired state of the ecosystem, can be informative as well (Figure 9; [236, 237]). In comparison to reference points, the challenge of achieving consensus on reference directions is small and can be applied in data-poor situations [233].

Figure 9. The relationship between target, benchmark, precautionary, and limit reference levels for an ecosystem indicator (adopted from [236]).



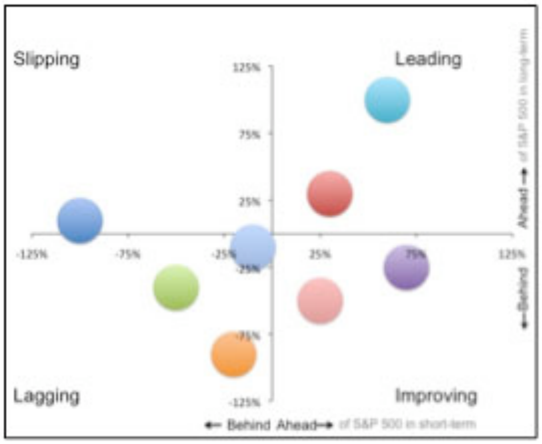
The concept of reference directions is familiar in the context of financial markets. For instance, the Dow Jones Industrial Average is an index representing the performance of 30 large, publicly owned U.S. corporations on the New York Stock Exchange. Though specific reference points are not widely agreed upon [238] it is generally accepted that increases in the Dow Jones are economically favorable and reductions are unfavorable.

A second financial market example illustrates an alternative approach for establishing reference directions, based on relative performance. The S&P 500, a weighted index consisting of 500 companies traded on the New York Stock Exchange, American Stock Exchange, and NASDAQ stock market [239], is commonly used to compare the direction of change of individual companies to the direction of change of the overall financial market ([240]; Figure 10). Companies that show greater percentage increases than the S&P 500 over the short-term (e.g., days, weeks, or months) and long-term (quarters or years) are considered to be leading the market, whereas companies that show lesser percentage increases than the S&P 500 over the short-term and long-term are considered to be lagging the market. Slipping companies are those that are behind the S&P 500 in the short-term but ahead in the long-term, and improving companies are those that are ahead of the S&P 500 in the short-term but behind in the long-term. This approach could be adopted for evaluating ecosystem indicators in Puget Sound relative to a summary index for each PSP goal, and would be useful for distinguishing indicators in need of management attention (lagging, slipping) from those on a desired trajectory (leading, improving).

Reference directions are already used widely in the management of natural systems. For instance, in San Francisco Bay and the North Sea increasing abundance of certain species of jellyfish is viewed as a sign of deteriorating ecosystem health [226], though no exact value corresponding to an undesired abundance level has been established. Similarly, a decline in disturbance-sensitive, specialist seabirds is viewed as indicative of strong anthropogenic influences (e.g., Chesapeake Bay; [241]) or worsening climatic conditions (e.g., central California coast; [242]), but a specific value for the rate or extent of decline marking an undesired state remains ambiguous. As a final example, in 2002 nearly 200 nations pledged to reduce the global rate of biodiversity loss by 2010 without establishing a target level for the amount of reduction that they desired [243].

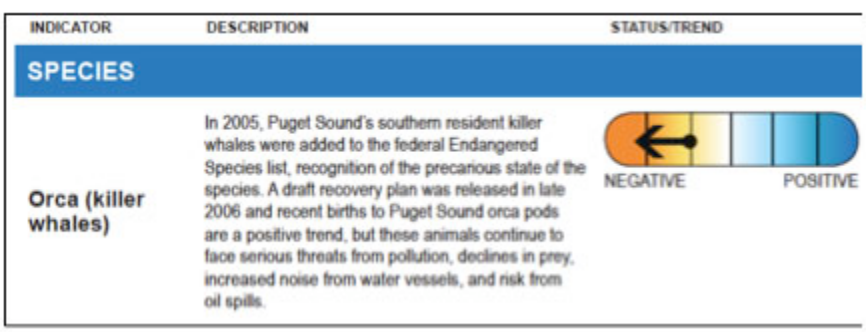
Figure 10. Use of reference directions based on relative performance of individual stocks (circles) and the S&P 500, a weighted index of overall market performance. Stocks that show greater percentage increases than the S&P 500 over the short-term (e.g., days, weeks, or months) and long-term (quarters or years) are considered to be leading the market, whereas stocks that show lesser percentage increases than the S&P 500 over the short-term and long-term are considered to be lagging the market. Slipping stocks are those that are behind the S&P 500 in the short-term

but ahead in the long-term, and improving stocks are those that are ahead of the S&P 500 in the short-term but behind in the long-term. Adapted from www.nytimes.com



In Puget Sound, reference directions for indicators could serve as placeholders in order to allow time for the development of more precise reference points. Indeed, the Puget Sound Action Team (PSAT) has applied the reference direction approach previously [244]. Using a simple and easily-interpreted schematic, PSAT evaluated indicators based on whether their status was generally negative, fair, or positive and whether the trend in the indicator was negative, neutral, positive, or unknown compared to a desired status (Figure 11). In future versions of the PSSU, a similar approach could be applied productively to the indicator assessments presented in Chapters 2 and 3, provided that the direction of change that is considered desirable for each indicator is specified explicitly and its rationale explained.

Figure 11. Example of indicator report card from the 2007 State of the Sound document. This figure shows that the status of one indicator of the health of Puget Sound species, orcas, is generally negative because the dot is to the left of center, and its trend, indicated by the arrow, is also negative. Reproduced from [244].



Target, benchmark, limit, and precautionary reference levels

A construct that has been particularly successful in the realm of fisheries management is the distinction between target and limit reference levels (Figure 9). A target is a reference level that signals a desired state, whereas a limit is a reference level pegged to an extreme value beyond which undesired change occurs [236, 245].

In fisheries and marine EBM limit reference levels thus identify what is to be avoided [20], and can be used to redirect and prioritize management action before irreversible harm occurs. Because of uncertainty inherent to the measurement of any indicator, precautionary or warning reference levels that are more conservative than the limit reference levels may be used (Figure 9; [236, 246]). Target reference levels identify what is to be achieved [20], and in so doing allow managers and policymakers to determine when their efforts and resource allocations have been sufficient [247]. Because indicators respond at varying rates to management actions, target reference levels may be most useful when accompanied by benchmarks, or indicator values suggestive of progress toward targets (Figure 9).

In Puget Sound, the PSP has taken it upon itself to establish targets and benchmarks. Because of legislated restoration and protection deadlines, the PSP has associated a timeline with target and benchmark reference levels. The PSP defines a target as a “desired future numeric value for an ecosystem status indicator in 2020.” Similarly, the PSP describes a benchmark as a “measurable interim (i.e., pre-2020) milestone set to demonstrate progress toward a target for an ecosystem status indicator” [76].

Importantly, the indicator associated with a target reference level need not be identical to the indicator associated with the corresponding benchmark. The current financial crisis provides a useful parallel to illustrate this point. The onset of the economic recession in the U.S. was characterized in part by a Gross Domestic Product (GDP) that fell for several months [239]. Thus a target reference level for economic recovery could be measured in terms of a consistent month-to-month rise in GDP. Benchmarks for measuring progress toward this target included a variety of indicators other than GDP, however, such as the number of new unemployment claims filed and new construction permits issued each week [248].

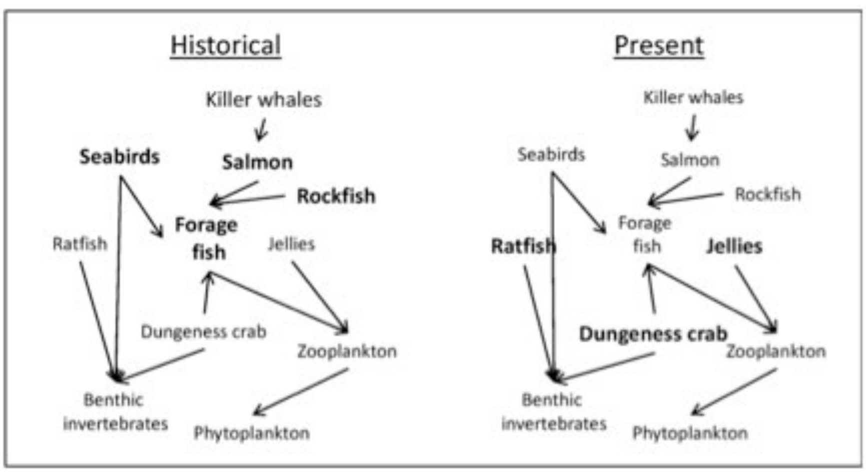
In the context of Puget Sound, a fundamental goal is to achieve a healthy and sustaining population of southern resident killer whales (SRKWs) [1], and one indicator of SRKW population status is the number of individuals in the population. The target reference level associated with the goal of SRKW population recovery may be measured using this indicator, but because the likely response time for achieving the target is several decades, a benchmark might be set using a different indicator, such as a reduced infant mortality rate or an increased annual population growth rate [249].

Because they are a primary interest of the PSP, we focus on approaches for determining target reference levels rather than limits. Though our discussion is framed largely in terms of reference points, we see no reason why targets cannot be defined in terms of reference directions, at least in the short term. However, it is not obvious how to distinguish a benchmark from a target using reference directions alone.

Baseline reference levels

Baseline reference levels are derived from time periods or locations free from human pressures. We use the term baseline inclusive of the structure and function of an ecosystem (1) prior to substantial human impact (i.e., during some 'baseline' time period [250, 251]), (2) inside of areas protected from human impacts [252, 253], and (3) in remote geographic locations subject to minimal human pressures [254]. Recognition of these types of reference levels is crucial for avoiding the shifting baselines syndrome—failing to identify the state of nature absent human impacts so that it is impossible to determine the extent of degradation [251]. As such, there is value in reconstructing time series of both desired and undesired changes in indicators, such as shifts in the abundances of iconic and nuisance species. It can also be quite useful to make comparisons across spatial locations that vary in the extent to which they have been altered by human activities [255]. Even where detailed information is not available, the qualitative difference between present and historic, or disturbed and undisturbed, values of ecosystem indicators can provide a reasonable starting point for determining target reference directions (Figure 12; [256]).

Figure 12. Comparison of a simplified historical and present-day Puget Sound marine food web. Larger, bold font indicates great erabundance/biomass. This figure is intended to be a conceptual schematic, and is not based on historical data. Historical and present-day could be replaced with unexploited and exploited areas or remote and metropolitan locations.



Historical information can be gleaned from a variety of sources, including paleo-ecological records [257] archaeological findings [258], historical documents [259, 260], and long-term ecological data [261, 262]. Additionally, interviews with people who have experience with an ecosystem during different eras of human impact can provide valuable insights into changes in ecosystem indicators over time [263, 264]. Indeed, subjective impressions of how indicators have varied through time can be standardized with known values and used to establish reference levels (e.g., unfished biomass of currently harvested species; [265]). One concern with using historical baselines, however, is that ecosystem dynamics are not necessarily stationary. Climatic shifts and other sources of variation can render historic states unattainable [236]. Such fundamental changes must be appreciated before making the decision to associate an indicator with a target reference level derived from a historical baseline.

Marine protected areas (MPAs) and areas with low human impact provide useful experiments for evaluating the natural biophysical state of an ecosystem absent major, direct anthropogenic influences [41]. Such spatial baseline ecosystems make particularly useful reference levels because they represent one extreme in a spectrum of management possibilities in the contemporary time period. Admittedly, problems exist with these approaches. For instance, geographic variability among reference and impacted sites and anthropogenic activities that manifest effects on regional and even global scales (e.g., climate change) can confound comparisons. Nonetheless, differences between indicators inside and outside of MPAs [266, 267] and near to and far from locations with high human population densities [268-271] can provide a useful basis for calibrating expectations regarding the healthy state of an ecosystem [254, 272, 273].

In Puget Sound, many untapped sources of baseline information exist. For example, archival papers document changes in the abundances of harvested species dating back to at least the 19th century [274]. According to these accounts, species declines appear to have occurred long ago, and quite rapidly: "[f]rom 1869 to 1877 it was not an uncommon occurrence for us to catch from 200 to 300 barrels of herring in a night, but since 1877... the largest night's work is about 20 barrels" [274]. Similarly, historical habitats have been altered drastically: <20% of tidal marshes present in the mid-19th century exist today [275]. Even shorter intervals reveal surprisingly large changes in ecosystem status: current concentrations of polybrominated diphenyl ethers (PBDEs) in southern resident killer whales dwarf the levels detected 10 years earlier [276]. In modern times, spatial differences in the ecological communities within and outside of marine reserves near Edmonds, Hood Canal, and the San Juan Islands suggest the direct negative impacts of fisheries on rockfishes and lingcod [277, 278]. Similarly, comparison of the most populated areas of Puget Sound to more rural areas reveals dramatic differences in the abundance of kelp [272, 279].

In terms of actually setting target and benchmark reference levels using information about baselines, the ultimate decision lies in the hands of policymakers [280]. Following on the example of the change analysis conducted for Puget Sound's tidal marshes, the question remains as to what target reference level is most appropriate given that >80% of the historic habitat has been destroyed since 1850. There is no single and absolutely correct answer to this question. It is up for negotiation among stakeholders, but the knowledge of what existed historically and/or what is currently observed in remote or protected locations provides an idea of what is possible.

Reference levels based on nonlinearities

Nonlinearities are common in nature [281, 282]. Sudden change in ecosystem attributes can result from seemingly smooth and gradual change in physical or biological components [283]. For instance, in kelp forests, increasing sea urchin densities initially produce small or negligible changes in habitat-providing kelp. However, above a threshold sea urchin density, declines in kelp and changes in associated ecological communities can be quite rapid [284, 285]. Similarly, on coral reefs, important ecosystem functions decline rapidly with initial increases in human impacts, but thereafter change quite slowly [254, 286]. These examples illustrate that nonlinearities in functional relationships distinguish environmental conditions or types of management actions leading to smooth and proportional changes in ecosystem state from those that cause abrupt and disproportionately large changes. An understanding of nonlinearities is highly relevant in the context of managing the Puget Sound ecosystem because it presents opportunities to define clear and objective reference points [287, 288].

Nonlinear functional relationships underpin commonly-used management reference points in fisheries and in the control of contaminants in the environment (e.g., chemicals, effluents, non-native species, etc.). For instance, the spawning stock biomass and the fishing mortality rate corresponding to maximum sustainable yield are two of many biological reference points used in single-species fisheries management [289]. The concept of maximum sustainable yield is based on the expectation that the yield from the fishery peaks at intermediate levels of population biomass and fishing mortality rate imposed on the target population. These nonlinear relationships are the consequence of assumptions in surplus production models of fish population dynamics, and make it possible to identify objectively a reference point on either side of which fishing yield is reduced. In ecotoxicology, contaminants frequently have little or no deleterious effects on biota below some minimum concentration but lead to serious sublethal or lethal effects thereafter (Figure 13 a,b). Thus, a reference point can be defined based on a threshold in such exposure-response relationships [232]. In both situations, the reference points are linked mathematically to a functional relationship of interest to managers and policymakers [246]. The functional relationships most relevant in a marine EBM context fall into two broad categories [281]. In both cases, the response variables of interest are ecosystem attributes that influence ecosystem health, and might include nutrient cycling, energetic rates, and resilience. These are akin to the toxin concentrations in ecotoxicological studies. In the first category, the predictor variable (analogous to the exposure effect in ecotoxicological studies) is some environmental condition(s). For example, reductions in the amount of upwelling along the west coast of the United States are associated with an exponential increase in seabird mortality events, which appear to be indicative of broader changes in ecosystem attributes, such as productivity [242]. In the second category, the predictor variable is a factor(s) under the control of managers and policymakers. For instance, a marine food web model for northern British Columbia suggests that several ecosystem attributes show nonlinear declines with increasing fishing pressure and with reductions in nearshore habitat quantity and quality [288]. In both cases, it is possible to define mathematically a point separating rapid and dramatic changes in the ecosystem attributes from more smooth and gradual changes (Figure 13c,d).

Reference levels for ecosystem indicators can be derived from either category of nonlinearity. The guidelines for selecting a reference point based on a functional relationship between predictable environmental conditions or factors under the control of managers and policymakers and ecosystem attributes are as follows:

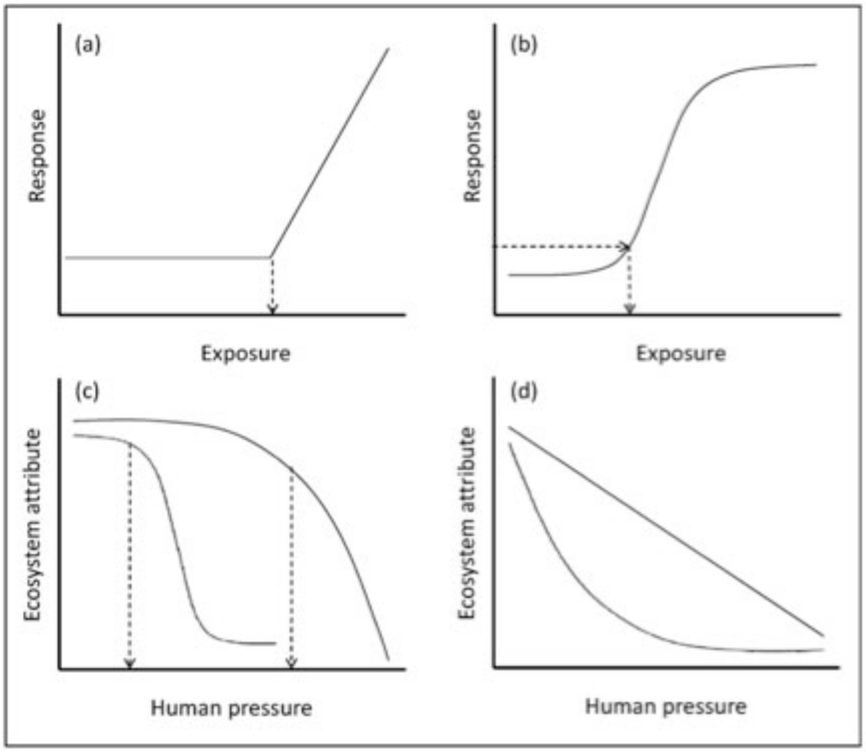
1. Examine the functional relationship of interest, using data, models, or both;
2. Use information theoretic techniques [290] to fit alternative linear and nonlinear mathematical functions to the relationship;
3. If the best-fit function is nonlinear, select a reference point that distinguishes the steep from the shallow portion of the curve [288].

Reasonable target reference levels for the sigmoidal and concave functional relationships shown in Figure 13c would correspond to portions of the curves where the value of the ecosystem attribute is high and the rate of change in the ecosystem attribute with increasing human pressure is low, i.e., where the dashed arrows intersect the curves.

The identification of nonlinear relationships between pressures and ecosystem attributes could be used productively to set target reference levels in Puget Sound. One way to detect nonlinearities relevant for food web health in particular would harness the power of a recently developed Ecopath model for the Central Basin of Puget Sound [26]. Indeed, Samhouri et al. (2010) recently followed the methods outlined in steps 1-3 above to determine food web reference levels associated with two different stressors (fishing and habitat modification) along the British Columbia coast [288]. Empirical examples of nonlinearities already exist as well. For instance, Rice (2007) found that there was a drastic and abrupt decline in the abundance of diving ducks and herons in Puget Sound above ~70% alongshore urban land cover [291]. Given the potential for these species to act as reliable indicators of ecosystem health [45, 118], a target reference level for their abundance based on the effects of urbanization may be sensible.

A concerted effort to gather information about functional relationships between ecosystem indicators and pressures would greatly advance efforts to set target and benchmark reference levels in Puget Sound. These reference points should be considered complementary to those based on baseline conditions.

Figure 13. Examples of nonlinear relationships in ecotoxicological (a-b) and ecosystem (c-d) studies. (a) A hockey stick relationship in which the reference point could be either the LOEC (lowest observed effect concentration), i.e., the lowest concentration causing an effect that is statistically different from control (upper 95% CI of x-axis threshold estimate), or a NOEC (no observed effect concentration), i.e., the highest concentration below LOEC (could be lower 95% CI of x-axis threshold estimate). (b) A sigmoidal relationship in which the reference point is an Ecp, the concentration causing the effect in proportion p of the population (e.g., LC50). (c) It is possible to identify objectively a reference point in terms of human pressure if the relationship between the predictor variable and the ecosystem attribute is sigmoidal or concave. (d) A convex relationship suggests that management actions that reduce human pressures to steeper portions of the function will produce the greatest improvements in the ecosystem attribute. Linear functions do not allow the objective identification of a threshold-based reference point. In all figures, dashed arrows indicate possible reference points. In (c) and (d), positive values on the y-axis are assumed to represent the desired state of the ecosystem attribute.



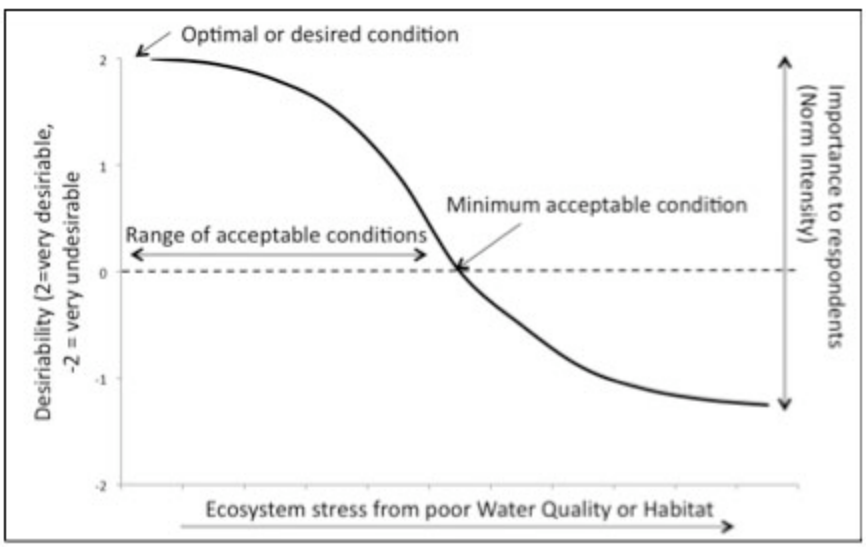
Normative reference levels

In the PSP parlance, a target is defined as a desired state [37]. Consequently, the process of establishing desirability must comprise not just ecological understanding, but also societal values [280, 292]. A powerful way to collect and organize data about societal values is the normative approach [293]. Norms define what is generally accepted within a cultural context, and may serve as societal standards to evaluate ecosystem conditions, human activities, or management strategies.

Norms are typically described by means of a graphic device referred to as a social norm curve (Figure 14; [294]). In applying this concept to ecosystem targets, the x-axis represents environmental stressors and the y-axis portrays stakeholder survey responses. Thus, social norm curves might represent the results from structured surveys in which respondents are asked about the acceptability of different ecosystem states, which vary with changes in pressures like water quality or habitat modification. The goal of stakeholder surveys is to identify the acceptability of alternative ecosystem scenarios that illustrate trade-offs among different aspects of ecosystem health (e.g., food web health, water quality, habitat, key species, and human well-being). Alternative scenarios can be portrayed using easily-interpreted, stylized artistic renderings of the ecosystem under consideration that highlight key trade-offs among different ecosystem components [295, 296]. Targets and benchmarks can be set based on scenarios that are deemed minimally acceptable by the average respondent, subject to legal, regulatory or other constraints. A key challenge with this approach is dealing with the fact ecosystem conditions are rarely produced by one individual's behavior but by the cumulative effects of many people's behavior.

In Puget Sound, the PSP and the World Resources Institute have already initiated the process of soliciting feedback from stakeholders about how they define a healthy Puget Sound [297]. This work could be built upon by extending social norms surveys to Native American tribes and stakeholder groups (e.g., commercial fishers, recreational fishers, agricultural interests, builders and developers, members of environmental organizations, coastal homeowners, etc.). In other marine systems around the world, similar surveys have been conducted by soliciting formal feedback about reference levels from regional scientists [298]. By establishing ranges of acceptability, the PSP can ensure that its targets are in sync with the desires of the public which they are meant to serve. Thus rigorously conducted normative surveys provide a tool to inform target selection within the realm of what is ecologically and legally possible and appropriate.

Figure 14. Hypothetical social norm curve. The x-axis shows increasing ecosystem stress from poor water quality or habitat, and the y-axis portrays stakeholder values regarding the desirability of different ecosystem states. Y-axis values >0 reflect socially acceptable ecosystem states, and the range of responses reflects the importance of ecosystem status to stakeholders.



Focus for the future: targets and success in Puget Sound

A catalog of ecosystem indicators is only useful in the extent to which it informs answers to the question "Is Puget Sound healthy?" In economics, it is not meaningful to report on the rate at which unemployment claims are filed unless it is known that an increase in that rate indicates a decline in the business cycle [248]. Similarly, in the absence of reference levels, a list of values for indicators alone provides no insight into the status of the ecosystem relative to its desired state. Thus, establishing a target associated with each indicator is fundamental to the success of the Puget Sound Partnership's ecosystem-based management efforts, for several reasons.

First, the articulation of targets associated with each indicator allows for a careful accounting of management successes and failures. Targets remove ambiguity from well-intended but vague policy goals and facilitate the development of a roadmap for new actions, policies, and management strategy evaluations. Pathways of ecosystem degradation may involve sequential losses of structural features (relative abundance of species), species, and functional components (all species responsible for particular ecological processes) [299]. Awareness of this type of progression can provide justification for benchmark reference levels that track recovery along similar pathways (but in reverse) toward more ambitious, longer-term targets.

Second, as described in the Futures section above, creating targets for individual indicators brings into focus the notion of trade-offs. For instance, interactions among species, such as harbor seals and forage fishes, may render obsolete target reference levels instituted for each group individually because some combinations of abundance are ecologically impossible. Likewise, establishing targets for contaminant loads related to water quality may interact with desired states of human well-being. The use of conceptual and quantitative ecosystem models and other tools can help to reveal the spectrum of possible combinations of target reference levels for multiple indicators simultaneously.

Third, target reference levels can also be viewed as the antecedent of legal statutes and regulations. In other words, the formal establishment of targets sets up a system of EBM accountability. These reference levels can be used as a springboard for enacting and enforcing policies to ensure that human activities do not exceed levels that would prevent the achievement of ecosystem recovery goals [300].

Fourth, targets can serve a useful role if they are linked to decision criteria or control rules [246, 287]. In other words, it would serve the PSP's interests if target values for indicators were associated with management responses. For instance, in the case of Chinook salmon in Puget Sound, achievement of the near-term recovery target of 1,600 spawners [15] might be linked to a control rule that influenced efforts to restore riparian vegetation and increase woody debris. Such built-in linkages would contribute to the efficient allocation of PSP financial resources and solidify a clear plan for active and adaptive management.

We have not yet attempted an exhaustive review of targets for each indicator evaluated in Section 4. A summary of existing targets specific to Puget Sound follows. For those indicators where targets or reference levels do not exist, it should be possible to determine appropriate targets using any of the three approaches outlined in Sections 5.5-5.7. Initially, it should suffice to define a reference direction for each indicator used to evaluate ecosystem status by identifying baselines, recognizing nonlinearities, or assessing social norms. Eventually, however, the PSP should strive to produce target reference points wherever possible. Key point: To be useful from a policy and management perspective, ecosystem indicators must be linked to reference levels. Reference levels provide context so that changes in indicator values can be interpreted relative to desired ecosystem states. Establishing targets for individual indicators brings into focus the notion of trade-offs among competing ecosystem services. The use of conceptual and quantitative ecosystem models can reveal the spectrum of possible combinations of target reference levels for multiple indicators simultaneously.

Existing Targets for Puget Sound

This section provides a brief summary of existing targets for Puget Sound including those for species, habitats, water quality, and water quantity.

Existing Species Targets

In Puget Sound, target reference levels have been assigned to a subset of ecosystem indicators. For indicators meant to inform the PSP Species Goal, it is worth noting that targets have been established primarily for species that have been listed as vulnerable, threatened, endangered, etc. at the state or federal level (especially marine mammals). Consequently, these targets frequently represent minimum requirements because many of the species were or are currently recovering from depressed states. Once achieved, such targets should be considered limit reference levels under the vocabulary introduced in this Section, and new targets should be established. Table 30 presents a selection of Species indicators that clearly met the “Linkable to scientifically-defined reference points and progress targets” criterion and for which targets have been defined in Puget Sound or Washington State specifically.

Existing Habitat Targets

We identified targets for two indicators meant to inform the PSP Habitats Goal: riparian habitat and aggregation/deposition zones (Table 31). For riparian habitats, we report targets for indicators intended to represent important ecosystem functions such as sediment, nutrient, and pollutant removal, erosion control, recruitment of large woody debris, regulated water temperature, availability of habitat for wildlife, and diversity of microclimates. For aggregation/deposition zones, we report a target that would ensure the maintenance of the structure and function of this habitat type in its current form.

Existing Water Quality Targets

The State of Washington has developed several sets of standards and criteria for both freshwater and marine surface water quality. Standards for physical and chemical parameters are generally established based on habitat type or water use category. For freshwater the Aquatic Life Use categories are summarized in Table 32; the Recreational Use categories are summarized in Table 36 [130, 138]. Water use designations for individual rivers and streams are listed by Water Resource Inventory Area (WRIA) in WAC 173-201A-602. The Aquatic Life Use categories for marine waters are summarized in Table 33. The majority of Puget Sound is listed as Extraordinary quality with the exception of designated bays and inlets (e.g. Elliot Bay, South Puget Sound, and Possession Sound) which are listed as either Excellent or Good. The sole area with a Poor designation is a potion of Commencement Bay, south and east of south 11th Street [301].

Summaries of the water quality criteria for physical and chemical properties in freshwater and marine water are presented in Table 32 and Table 33, respectively. Nutrient action levels for lakes are listed in Table 34. Surface water quality criteria for freshwater and marine waters for trace organic and inorganic chemicals is shown in Table 35; additional criteria for the protection of human health are included in Chapter 40 of the Code of Federal Regulations [302]. Water quality criteria for bacteria, which are meant to be protective of human health, are listed in Table 36.

Existing Water Quantity Targets

There are three indicators of Freshwater Water Quantity with established goals or targets (Table 37). Instream flow rule establish minimum flow requirements on several rivers and streams in the Puget Sound region. The flow rules are meant to legally acknowledge ecological flow requirements. A detailed review of the actual flow regimes versus the instream flow rules is presented in Chapter 2 of the PSSU.

There are also targets for flooding that are established at each gauge station. While not strictly goals, these can be used to monitor the potentially effects of land use change or climate change on flooding. Finally the State of Washington has established efficiency requirements through the Municipal Water Law. While this does not strictly define conservation targets it does mandate system loss limited and the establishment of efficiency programs within each supply system.

Tables - Defining ecosystem reference levels

Table 30. Species indicators for which targets have been established in Puget Sound and/or Washington state.

Species indicator	Target	Achieved	Reference
Bald eagle	Equilibrium population abundance is ~6,000 individuals in WA state	Yes	[303]
Harbor seal	Carrying capacity of 10,000-13,000 individuals (WA inland waters)	Yes	[304]
Peregrine falcon	Delisting criteria: 30 reproductive pairs in WA state; 1.5 young/territorial pair per year for a 5-year period	Yes	[305]
Pinto abalone	Achieve >0.15 individuals m^{-2} to avoid Allee effects due to reproductive failure	No	[117]
Southern Resident killer whale	Delisting criteria: Sustained average population growth of 2.3% per year for 28 yrs	No	[249]

Table 31. Habitat indicators for which targets have been established in Puget Sound and/or Washington state.

Habitat indicator	Target	Reference
Riparian habitat		
% of riparian habitat with a lateral extent >30 m	70-80%	[306]
% of riparian habitat with a lateral extent >100 m	40-50%	[306]
% of riparian habitat with a lateral extent <10m (encroachment)	10-20%	[306]
Corridor continuity (road crossings/km)	1-2	[306]
% natural forest or wetland cover	75-90%	[306]
% mature native vegetation or wetland	75-90%	[306]
Buffer width for 75% effective nitrogen removal	25m	[307]
Aggregation/deposition zones		
Deposition rate	>0.32 cm yr^{-1}	[328-331]

Table 32. Freshwater water quality criteria per Washington Administrative Code based on aquatic life use

Parameter	Categories for Freshwater Aquatic Life								Ref.	Notes
	River - Char Spawning and Rearing	River - Core Summer Salmonid Habitat	River - Salmonid Spawning, Rearing, and	River - Salmonid Rearing and Migration Only	River - Non-anadromous Interior Redband Trout	River - Indigenous Warm Water Species	River - Other	Lakes		
Dissolved Oxygen (Lowest 1-Day Min.)	9.5 mg/L	9.5 mg/L	8.0 mg/L	6.5 mg/L	8.0 mg/L	6.5 mg/L	Not less than 0.2 mg/L below natural conditions.		[92, 130]	
Temperature 7-day average of the daily max. temp. (7-DADMax)	12°C (53.6°F)	16°C (60.8°F)	17.5°C (63.5°F)	17.5°C (63.5°F)	18°C (64.4°F)	20°C (68°F)	Not more than 0.3°C above natural conditions.		[93, 130, 308]	a)
pH	6.5 - 8.5, human-caused variation less than 0.2 units		6.5 - 8.5; human-caused variation less than 0.5 units						[130]	b)
Total Dissolved Gas	Less than 110 % of saturation								[130]	
Turbidity	5 NTU increase when the background is 50 NTU or less; or A 10% increase when the background is 50 NTU or more.			10 NTU (<50 NTU); 20% inc. (>50 NTU)	5 NTU (<50 NTU); 10% inc. (>50 NTU)	10 NTU (<50 NTU); 20% inc. (>50 NTU)			[130]	
Notes: a) Special protection per Ecology publication 06-10-038. 7-DADMax = 9°C (48.2°F) at the initiation of spawning and at fry emergence for char. 7-DADMax = 13°C (55.4°F) at the initiation of spawning for salmon and at fry emergence for salmon and trout. b) See [309] for a non-compliance analysis										

Table 33 - Marine water quality criteria per Washington Administrative Code based on aquatic life use.

Parameter	Categories for Marine Water Aquatic Life					Ref.	Notes
	Extraordinary quality	Excellent quality	Good quality	Fair quality	Other		
Dissolved Oxygen (Lowest 1-Day Min.)	7.0 mg/L	6.0 mg/L	5.0 mg/L	4.0 mg/L	Not less than 0.2 mg/L: below natural conditions	[92, 131]	
Temperature (1-day max.)	13°C (55.4°F)	16°C (60.8°F)	19°C (66.2°F)	22°C (71.6°F)	Not more than 0.3°C above natural conditions	[93, 131]	a)
pH	7-8.5; human-caused var. <0.2 units	7-8.5; human-caused var. <0.5 units		6.5-9.0; human-caused var. <0.5 units		[131]	
Total Dissolved Gas							
Turbidity	• 5 NTU increase when background is 50 NTU or less; or • A 10% increase in when the background is 50 NTU or more.		• 10 NTU increase when background is 50 NTU or less; or • A 20% increase in when the background is 50 NTU or more.				
Notes: a) Criteria for Other marine water is based on 7-day average of maximum daily temperature (7-DADMax)							

Table 34. Nutrient action levels for lakes in the Puget Sound ecoregion. If epilimnetic TP values exceed action levels a lake-specific study should be implemented per WAC 173-201A-230 (2).

Trophic State	Ambient Total Phosphorus (μg/l)	Criteria	Ref.
Ultra-oligotrophic	0-4	<4	[136, 308]
Oligotrophic	4-10	<10	[136, 308]
Lower mesotrophic	10-20	<20	[136, 308]
Action Value	>20		[136, 308]

Table 35. Water quality criteria for toxic substances for the protection of aquatic life. For human health standards see 40CFR Ch.1 (7–1–06 Edition) 131.36. References: [302, 310]

	Freshwater		Marine Water	
Substance	**Acute[a]**	**Chronic[b]**	**Acute[a]**	**Chronic[b]**
Aldrin/Dieldrin[e]	2.5	0.0019	0.71	0.0019
Ammonia	f,c	g,d	0.233[h,c]	0.035[h,d]
(unionized NH_3)[hh]	g	g		
Arsenic[dd]	360.0[c]	190.0[d]	69.0[c,ll]	36.0[d,cc,ll]
Cadmium[dd]	i,c	j,d	42.0[c]	9.3[d]
Chlordane	2.4	0.0043	0.09	0.004
Chloride (Dissolved)[k]	860.0[h,c]	230.0[h,d]	-	-
Chlorine (Total Residual)	19.0[c]	11.0[d]	13.0[c]	7.5[d]
Chlorpyrifos	0.083[c]	0.041[d]	0.011[c]	0.0056[d]
Chromium (Hex)[dd]	15.0[c,l,ii]	10.0[d,jj]	1,100.0[c,l,ll]	50.0[d,ll]
Chromium (Tri)[gg]	m,c	n,d	-	-
Copper[dd]	o,c	p,d	4.8[c,ll]	3.1[d,ll]
Cyanide[ee]	22.0[c]	5.2[d]	1.0[c,mm]	d,mm
DDT (and metabolites)	1.1	0.001	0.13	0.001
Dieldrin/Aldrin[e]	2.5	0.0019	0.71	0.0019
Endosulfan	0.22	0.056	0.034	0.0087
Endrin	0.18	0.0023	0.037	0.0023
Heptachlor	0.52	0.0038	0.053	0.0036
Hexachlorocyclohexane (Lindane)	2.0	0.08	0.16	-
Lead[dd]	q,c	r,d	210.0[c,ll]	8.1[d,ll]
Mercury[s]	2.1[c,kk,dd]	0.012[d,ff]	1.8[c,ll,dd]	0.025[d,ff]
Nickel[dd]	t,c	u,d	74.0[c,ll]	8.2[d,ll]
Parathion	0.065[c]	0.013[d]	-	-
Pentachlorophenol (PCP)	w,c	v,d	13.0[c]	7.9[d]
Polychlorinated				
Biphenyls (PCBs)	2.0	0.014	10.0	0.030
Selenium	20.0[c,ff]	5.0[d,ff]	290[c,ll,dd]	71.0[d,x,ll,dd]
Silver[dd]	y	-	1.9[ll]	-
Toxaphene	0.73[c,z]	0.0002[d]	0.21[c,z]	0.0002[d]
Zinc[dd]	aa,c	bb,d	90.0[c,ll]	81.0[d,ll]

NOTES:	
a.	An instantaneous concentration not to be exceeded.
b.	A 24-hour average not to be exceeded.
c.	A 1-hour average concentration not to be exceeded more than once every three years on the average.
d.	A 4-day average concentration not to be exceeded more than once every three years on the average.
e.	Sum of the Aldrin and Dieldrin concentrations.
f.	Shall not exceed the numerical value in total ammonia (mg N/L) given by:

For salmonids present:	0.275	+	39.0
	$1 + 10^{7.204-pH}$		$1 + 10^{pH-7.204}$
For salmonids absent:	0.411	+	58.4
	$1 + 10^{7.204-pH}$		$1 + 10^{pH-7.204}$

g.	Shall not exceed the numerical concentration calculated as follows:
	Unionized ammonia concentration for waters where salmonid habitat is an existing or designated use:

0.80 ÷ (FT)(FPH)(RATIO)			
where:	FT	=	1.4; for $15 \le T \le 30$
	FT	=	$10^{[0.03(20-T)]}$; for $0 \le T \le 15$
	FPH	=	1; for $8 \le pH \le 9$
	FPH	=	$(1 + 10^{(7.4-pH)}) \div 1.25$; for $6.5 \le pH \le 8.0$
	RATIO	=	13.5; for $7.7 \le pH \le 9$
	RATIO	=	$(20.25 \times 10^{(7.7-pH)}) \div (1 + 10^{(7.4-pH)})$; for $6.5 \le pH \le 7.7$

	Total ammonia concentrations for waters where salmonid habitat is not an existing or designated use and other fish early life stages are absent:
	where: A = the greater of either T (°C) or 7.
	Applied as a thirty-day average concentration of total ammonia nitrogen (in mg N/L) not to be exceeded more than once every three years on average. The highest four-day average within the thirty-day period should not exceed 2.5 times the chronic criterion.
	Total ammonia concentration for waters where salmonid habitat is not an existing or designated use and other fish early life stages are present:
	where: B = the lower of either 2.85, or $1.45 \times 10^{0.028 \times (25-T)}$. T = temperature °C.
	Applied as a thirty-day average concentration of total ammonia nitrogen (in mg N/L) not to be exceeded more than once every three years on the average. The highest four-day average within the thirty-day period should not exceed 2.5 times the chronic criterion.
h.	Measured in mg/L.
i.	≤ (0.944)(e(1.128[ln(hardness)]-3.828)) at hardness = 100. Conversion factor (CF) of 0.944 is hardness dependent. CF is calculated for other hardnesses as follows: CF = 1.136672 - [(ln hardness)(0.041838)].
j.	≤ (0.909)(e(0.7852[ln(hardness)]-3.490)) at hardness = 100. Conversions factor (CF) of 0.909 is hardness dependent. CF is calculated for other hardnesses as follows: CF = 1.101672 - [(ln hardness)(0.041838)].
k.	Criterion based on dissolved chloride in association with sodium. This criterion probably will not be adequately protective when the chloride is associated with potassium, calcium, or magnesium, rather than sodium.
l.	Salinity dependent effects. At low salinity the 1-hour average may not be sufficiently protective.
m.	$\le (0.316)e^{[0.8190[\ln(hardness)] + 3.688]}$
n.	$\le (0.860)e^{[0.8190[\ln(hardness)] + 1.561]}$
o.	$\le (0.960)(e^{[0.9422[\ln(hardness)] - 1.464]})$
p.	$\le (0.960)(e^{[0.8545[\ln(hardness)] - 1.465]})$
q.	$\le (0.791)(e^{[1.273[\ln(hardness)] - 1.460]})$ at hardness = 100. Conversion factor (CF) of 0.791 is hardness dependent. CF is calculated for other hardnesses as follows: CF = 1.46203 - [(ln hardness)(0.145712)].
r.	$\le (0.791)(e^{[1.273[\ln(hardness)] - 4.705]})$ at hardness = 100. Conversion factor (CF) of 0.791 is hardness dependent. CF is calculated for other hardnesses as follows: CF = 1.46203 - [(ln hardness)(0.145712)].
s.	If the four-day average chronic concentration is exceeded more than once in a three-year period, the edible portion of the consumed species should be analyzed. Said edible tissue concentrations shall not be allowed to exceed 1.0 mg/kg of methylmercury.
t.	$\le (0.998)(e^{[0.8460[\ln(hardness)] + 3.3612]})$
u.	$\le (0.997)(e^{[0.8460[\ln(hardness)] + 1.1645]})$
v.	$\le e^{[1.005(pH) - 5.290]}$
w.	$\le e^{[1.005(pH) - 4.830]}$
x.	The status of the fish community should be monitored whenever the concentration of selenium exceeds 5.0 ug/ l in salt water.
y.	$\le (0.85)(e^{[1.72[\ln(hardness)] - 6.52]})$
z.	Channel Catfish may be more acutely sensitive.
aa.	$\le (0.978)(e^{[0.8473[\ln(hardness)] + 0.8604]})$
bb.	$\le (0.986)(e^{[0.8473[\ln(hardness)] + 0.7614]})$
cc.	Nonlethal effects (growth, C-14 uptake, and chlorophyll production) to diatoms (*Thalassiosira aestivalis* and *Skeletonema costatum*) which are common to Washington's waters have been noted at levels below the established criteria. The importance of these effects to the diatom populations and the aquatic system is sufficiently in question to persuade the state to adopt the USEPA National Criteria value (36 µg/L) as the state threshold criteria, however, wherever practical the ambient concentrations should not be allowed to exceed a chronic marine concentration of 21 µg/L.
dd.	These ambient criteria in the table are for the dissolved fraction. The cyanide criteria are based on the weak acid dissociable method. The metals criteria may not be used to calculate total

Table 36. Bacteria water quality standards for Freshwater and Marine Water by water use category as defined by the Washington Administrative Code.

Parameter	Water Use Category				
	Extraordinary Primary Contact Recreation	**Primary Contact Recreation**	**Secondary Contact Recreation**	**Shellfish Harvest**	**Ref.**
Freshwater					
Fecal Coliform (geometric mean)	50 col./100 mL	100 col./100 mL	200 col./100 mL		[130]
Fecal Coliform (maximum)	100 col./100 mL	200 col./100 mL	400 col./100mL		[130]
Marine Water					
Fecal Coliform (geometric mean)		14 col./100 mL	70 col./100 mL	14 col./100 mL	[131]
Fecal Coliform (maximum)		43 col./100 mL	208 col./100 mL	43 col./100 mL	[131]

Table 37. Water Quantity indicators for which targets have been established in Puget Sound and/or Washington state.

Water Quantity indicator	Target	Achieved	Reference
Instream Flow Rules	Instream flow rules have been established for several streams and rivers in the Puget Sound watershed.	No[1]	[311-313]
Flood Stage	River Flood Stage	No	[314]
Per Capita Water Use	Municipal Water Law requires efficiency programs for suppliers	Yes	[315]
Notes - see PSSU Chapter 2a, Status and Trends of Violations of Instream Flow Rules			

Glossary

Attribute	characteristic that is of scientific and/or management importance, but insufficiently specific and/or logistically challenging to measure directly; also, ecological characteristic that specifically describes the state of Focal Components
Baseline	reference level derived from time periods or locations free from human pressures
Benchmark	indicator value suggestive of progress toward targets
CCME	Canadian Council of Ministers of the Environment
CFR	Code of Federal Regulations
Criteria	standards against which indicators were evaluated
Data considerations	indicator evaluation criteria related to the actual measurement of the indicator
DO	Dissolved Oxygen
Domain	distinct ecological areas that contain unique qualities or traits; terrestrial, freshwater, marine, interface/ecotone
Driver	factor that result in pressures that cause changes in the system
Driver-Pressure-State-Impact-Response (DPSIR)	conceptual framework that has been broadly applied in terrestrial and aquatic environmental assessments
EBM	Ecosystem Based Management
Ecosystem assessment indicator	technically robust and rigorous metric used by scientists and managers to understand of ecosystem structure and function
EPM	Ecosystem Portfolio Model
ESA	Endangered Species Act
Focal component	the major ecological characteristics of an ecosystem that capture the relevant scientific information in a limited number of discrete, but not necessarily independent categories
FRAP	Future Risk Assessment Project
GDP	Gross Domestic Product
GIS	Geographical Information System
Impact	measures of the effect of change in state variables such as loss of biodiversity, declines in productivity and yield, etc
Improving indicator	indicator that is increasing faster in the short-term but slower in the long-term than an index that captures aggregate changes in multiple indicators
IBI	Index of Biologic Integrity

Indicator	quantitative biological, chemical, physical, social, or economic measurements that serve as proxies for difficult-to-measure attributes of natural and socio-economic systems
JISAO	Joint Institute for the Study of the Atmosphere and Ocean
Lagging indicator	indicator that is increasing slower in the short- and long-term than an index that captures aggregate changes in multiple indicators
Leading indicator	indicator that is increasing faster in the short- and long-term than an index that captures aggregate changes in multiple indicators
Limit	reference level pegged to an extreme value beyond which undesired change occurs
Management strategy evaluation (MSE)	conceptual framework that enables the testing and comparison of different management strategies designed to achieve specified management goals
MPA	Marine protected areas
NMFS	NOAA National Martine Fisheries Service
Nonlinearity	sudden change in a response variable resulting from smooth and gradual change in a causal factor
Normative reference level	reference level defined based on what is socially acceptable, i.e., according to norms
Norms	define what is generally accepted within a cultural context, and may serve as societal standards to evaluate ecosystem conditions, human activities, or management strategies
Open Standards	Open Standards for the Practice of Conservation, developed by the Conservation Measures Partnership, Version 2.0 released in 2007. Available at http://www.conservationmeasures.org/initiatives/standards-for-project-management. The Open Standards are a series of five steps that comprise the project management cycle, with the aim of providing a framework and guidance for successful conservation action. They define conservation efforts as "projects," and bring together common concepts, approaches, and terminology in conservation project design, management and monitoring. For more information, see [3].
Other considerations	indicator evaluation criteria that make an indicator useful, but without which an indicator remains scientifically informative
PAH	polycyclic aromatic hydrocarbons
PBT	Persistent Bioaccumulative Toxics
PCB	polychlorinated biphenyls
PDBE	polybrominated diphenyl ethers
Performance Management	A system to track implementation and communicate progress of a conservation project or program

Precautionary reference level	reference level pegged to an extreme value beyond which undesired change occurs, but set to be more conservative than the limit; a.k.a. warning reference level
Pressure	factor that cause changes in state or condition. They can be mapped to specific drivers
Primary considerations	essential indicator evaluation criteria that should be fulfilled by an indicator in order for it to provide scientifically useful information about the status of the ecosystem in relation to PSP goals
PSAT	Puget Sound Action Team
PSNERP	Puget Sound Nearshore Ecosystem Restoration Project
PSP	Puget Sound Partnership
PSP Goals	combine societal values and scientific understanding to define a desired ecosystem condition, and include: Human health, Human well-being, Species and Food Webs, Habitats, Water Quantity, Water Quality
PSSU	Puget Sound Science Update
Ranking scheme	approach used to weight indicator evaluation criteria
Reference direction	which specifies how the trend in an indicator relates to the desired state of the ecosystem
Reference level	Point value or direction of change used to provide context so that changes in indicator values can be interpreted relative to desired ecosystem states
Reference point	Precise values of indicators used to provide context for the current status of an indicator
Response	Actions (regulatory and otherwise) that are taken in response to predicted impacts
Results chains	Map specific management strategies to their expected outcome (e.g., reduction of a threat) and their impact on key components of the ecosystem. One component in the Open Standards framework being used by the PSP to guide its performance management strategy. Results chains are diagrams that show how a particular action taken will lead to some desired result, by linking short-, medium- and long-term results in "if...then" statements. Comprised of three basic elements: strategy, expected outcomes, and desired impacts. Developed for use as part of the Puget Sound Partnership's Performance Management System in {Neuman, 2009 #20}.
Slipping indicator	Indicator that is increasing faster in the long-term but slower in the short-term than an index that captures aggregate changes in multiple indicators
SMA	Shoreline Management Act
SRKW	southern resident killer whales
State	Condition of the ecosystem (including physical, chemical, and biotic factors)
Target	Reference level that signals a desired state
Threats	Any activities that have altered the ecosystem in the past or present, or are

	likely to in the future
UERL	Urban Ecology Research Lab
USFWS	U.S. Fish and Wildlife Service
Vital sign indicator	Scientifically meaningful, but simple, metric that can generally inform the public and policy makers about the state of the ecosystem
WAC	Washington Administrative Code
WDFW	Washington Department of Fish and Wildlife
WDNR	Washington Department of Natural Resources
WDOE	Washington Department of Ecology
WDOH	Washington Department of Health
WQI	Water Quality Index

References

1. Puget Sound Partnership, Puget Sound Action Agenda, Protecting and Restoring the Puget Sound Ecosystem by 2020. 2008: Olympia.

2. Ruckelshaus, M., T.E. Essington, and P.S. Levin, Puget Sound, Washington, U.S.A., in Ecosystem-based management for the oceans, K.L. McLeod and H. Leslie, Editors. 2009, Island Press: Washington, D.C. p. 201-226.

3. Conservation Measures Partnership, Open Standards for the Practice of Conservation, Version 2.0. 2007.

4. Puget Sound Partnership, Using Results Chains to Develop Objectives and Performance Measures for the 2008 Action Agenda, P.S. Partnership, Editor. 2009.

5. Rapport, D.J., H.A. Regier, and T.C. Hutchinson, Ecosystem Behavior Under Stress. American Naturalist, 1985. 125(5): p. 617-640.

6. De Leo, G.A. and S. Levin, The multifaceted aspects of ecosystem integrity. Conservation Ecology, 1997. 1(3).

7. Levin, S.A., ORCHESTRATING ENVIRONMENTAL-RESEARCH AND ASSESSMENT. Ecological Applications, 1992. 2(2): p. 103-106.

8. Orians, G.H. and D. Policansky, Scientific Bases of Macroenvironmental Indicators. Annual Review of Environment and Resources, 2009. 34: p. 375-404.

9. Puget Sound Regional Council, Vision 2040. 2009: Seattle, WA. p. 144.

10. Puget Sound Regional Council, Regional Economic Strategy for the Central Puget Sound Region. 2005: Seattle, WA. p. 90.

11. Urban Ecology Research Lab, Puget Sound Future Scenarios. 2009: Seattle, WA.

12. Reid, W.V., et al., eds. Bridging Scales and Knowledge Systems: Concepts and Applications in Ecosystem Assessment. 2006, Island Press: Washington. 353.

13. Labiosa, W.B., et al., The South Florid Ecosystem Portfolio Model - A Map-Based Multicriteria Ecological, Economic, and Community Land-use Planning Tool, U.S.G. Survey, Editor. 2009: Menlo Park, CA. p. 41.

14. Bolte, J., Envision: A spatially explicity, distributed, multiagent-based framework for policy assessment and alternative futuring. 2009: Corvallis, OR.

15. Shared Strategy for the Puget Sound, Puget Sound Salmon Recovery Plan. 2007: Seattle, WA.

16. Boesch, D.F. and J. Greer, Chesapeake Futures: Choices for the 21st Century. 2003, Chesapeake Research Consortium, Inc.: Edgewater, MD.

17. Roberts, A.D., et al., Effects of projected future land cover on nitrogen and phosphorus runoff to Chesapeake Bay. Ecological Engineering, 2009. 35: p. 1758-1772.

18. Louisiana-Mississippe Gulf Couast Ecosystem Restoration Working Group, Roadmap for Restoring Ecosystem Resiliency and Sustainability. 2010: Washington, D.C.

19. American Association of Port Authorities. World Port Ranking. 2008 [cited 2010 March 11, 2010]; Available from: http://aapa.files.cms-plus.com/Statistics/WORLD_PORT_RANKINGS_20081.pdf.

20. Sainsbury, K.J., A.E. Punt, and A.D.M. Smith, Design of operational management strategies for achieving fishery ecosystem objectives. ICES Journal of Marine Science, 2000. 57(3): p. 731-741.

21. Walters, C., Adaptive management of renewable resources. 1986, New York: MacMillan.

22. Magnusson, K. and G. Stefansson, A feedback strategy to regulate catches from a whale stock. Reports of the International Whaling Commission 1989(Special Issue 11): p. 171-189.

23. Wiley, M.W. and R.N. Palmer, Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce, 2008. 134(3): p. 239-246.

24. Roberts, M., J. Newton, and D. Hannafious, Quality assurance project plan: Hood Canal Dissolved Oxygen Program integrates assessment and modeling study, year 1 activities., W.D.o. Ecology, Editor. 2005: Olympia, WA. p. 100.

25. Hepinstall, J.A., M. Alberti, and J.M. Marzluff, Predicting land cover change and avian community responses in rapidly urbanizing environments. Landscape Ecology, 2008. 23: p. 1257-1276.

26. Harvey, C.J., et al., A mass-balance model for evaluating food web structure and community-scale indicators in the Central Basin of Puget Sound. in review. NMFS-NWFSC-XXX.

27. Cocker Fennessy, Opinion Research and Outreach Synopsis. A Report for the Puget Sound Partnership. 2006: Seattle, WA.

28. U.S. Environmental Protection Agency, EPA's 2008 Report on the Environment. 2008, National Center for Environmental Assessment, Washington, DC; EPA/600/R-07/045F.

29. Kurtz, J.C., L.E. Jackson, and W.S. Fisher, Strategies for evaluating indicators based on guidelines from the Environmental Protection Agency's Office of Research and Development. Ecological Indicators, 2001. 1(1): p. 49-60.

30. Landres, P.B., J. Verner, and J.W. Thomas, ECOLOGICAL USES OF VERTEBRATE INDICATOR SPECIES - A CRITIQUE. Conservation Biology, 1988. 2(4): p. 316-328.

31. Fleishman, E. and D.D. Murphy, A Realistic Assessment of the Indicator Potential of Butterflies and Other Charismatic Taxonomic Groups. Conservation Biology, 2009. 23(5): p. 1109-1116.

32. Levin, P., et al., Integrated ecosystem assessments. 2008, National Oceanic and Atmospheric Administration: Seattle, WA.

33. Niemeijer, D. and R.S. de Groot, A conceptual framework for selecting environmental indicator sets. Ecological Indicators, 2008. 8(1): p. 14-25.

34. O'Neill, S.M., C.F. Bravo, and T.K. Collier, Environmental Indicators for the Puget Sound Partnership: A Regional Effort to Select Provisional Indicators (Phase 1). Summary Report. 2008, National Oceanic and Atmospheric Administration: Seattle, WA.

35. Puget Sound Ambient Monitoring Program, A conceptual model for environmental monitoring of a marine system. 2000, Washington Department of Fish and Wildlife.

36. U.S. Environmental Protection Agency. Puget Sound Georgia Basin Ecosystem Indicators. 2010; Available from: http://www.epa.gov/region10/psgb/indicators/index.htm.

37. Puget Sound Partnership, Identification of Ecosystem Components and Their Indicators and Targets. 2009, Puget Sound Partnership.

38. Washington Department of Fish and Wildlife. Puget Sound Ambient Monitoring Program. 2010; Available from: http://wdfw.wa.gov/fish/psamp/study.htm.

39. Puget Sound Nearshore Ecosystem Restoration Project, Conceptual model for assessing restoration of Puget Sound nearshore ecosystem. 2006, University of Washington.: Seattle, WA.

40. Puget Sound Action Team, 2007 Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. 2007: Olympia. p. 260.

41. Rice, J.C. and M.-J. Rochet, A framework for selecting a suite of indicators for fisheries management. ICES J. Mar. Sci., 2005. 62(3): p. 516-527.

42. U.S. Environmental Protection Agency, A Framework for Assessing and Reporting on Ecological Condition: A Science Advisory Board Report, T.F. Young and S. Sanzone, Editors. 2002: Washington, D.C.

43. Harwell, M.A., et al., A framework for an ecosystem integrity report card. Bioscience, 1999. 49(7): p. 543-556.

44. Heinz Center, The State of the Nation's Ecosystems 2008: Measuring the Lands, Waters, and Living Resources of the United States. 2008: Island Press.

45. Fulton, E.A., A.D.M. Smith, and A.E. Punt, Which ecological indicators can robustly detect effects of fishing? ICES Journal of Marine Science, 2005. 62(3): p. 540-551.

46. Niemi, G.J. and M.E. McDonald, Application of ecological indicators. Annual Review of Ecology Evolution and Systematics, 2004. 35: p. 89-111.

47. Soule, M.E., et al., Strongly interacting species. conservation policy, management, and ethics. Bioscience, 2005. 55(2): p. 168-176.

48. Heiman, K., Strongly interacting nonindigenous species. Bioscience, 2005. 55(7): p. 548-548.

49. Wikelski, M. and S.J. Cooke, Conservation physiology. Trends in Ecology & Evolution, 2006. 21(1): p. 38-46.

50. Wikelski, M., L.M. Romero, and H.L. Snell, Marine Iguanas Oiled in the Galapagos. Science, 2001. 292(5516): p. 437-438.

51. Marty, G.D., et al., Role of disease in abundance of a Pacific herring (Clupea pallasi) population. Canadian Journal of Fisheries and Aquatic Sciences, 2003. 60(10): p. 1258-1265.

52. Landis, W.G. and P.T. Bryant, Using Weight of Evidence Characterization and Modeling to Investigate the Cause of the Changes in Pacific Herring (Clupea pallasi) Population Dynamics in Puget Sound and at Cherry Point, Washington. Risk Analysis, 2010. 30(2): p. 183-202.

53. Rowe, C.L., "The Calamity of So Long Life": Life histories, contaminants, and potential emerging threats to long-lived vertebrates. Bioscience, 2008. 58(7): p. 623-631.

54. Beldomenico, P.M. and M. Begon, Disease spread, susceptibility and infection intensity: vicious circles? Trends in Ecology & Evolution, 2010. 25(1): p. 21-27.

55. Arkoosh, M.R., et al., Effect of pollution on fish diseases: Potential impacts on salmonid populations. Journal of Aquatic Animal Health, 1998. 10(2): p. 182-190.

56. Baker, M.R. and D.E. Schindler, Unaccounted mortality in salmon fisheries: non-retention in gillnets and effects on estimates of spawners. Journal of Applied Ecology, 2009. 46(4): p. 752-761.

57. Hsieh, C.H., et al., Fishing effects on age and spatial structures undermine population stability of fishes. Aquatic Sciences, 2010. 72(2): p. 165-178.

58. Francis, R.C., et al., Fisheries management - Ten commandments for ecosystem-based fisheries scientists. Fisheries, 2007. 32(5): p. 217-233.

59. Holmes, E.E., et al., Age-structured modeling reveals long-term declines in the natality of western Steller sea lions. Ecological Applications, 2007. 17(8): p. 2214-2232.

60. Schwartz, M.K., G. Luikart, and R.S. Waples, Genetic monitoring as a promising tool for conservation and management. Trends in Ecology & Evolution, 2007. 22(1): p. 25-33.

61. Levin, P.S. and M.S. Schiewe, Preserving salmon biodiversity. American Scientist, 2001. 89: p. 220-228.

62. Allendorf, F.W., et al., Genetic effects of harvest on wild animal populations. Trends in Ecology & Evolution, 2008. 23(6): p. 327-337.

63. Westemeier, R.L., et al., Tracking the long-term decline and recovery of an isolated population. Science, 1998. 282(5394): p. 1695-1698.

64. Walsh, M.R., et al., Maladaptive changes in multiple traits caused by fishing: impediments to population recovery. Ecology Letters, 2006. 9(2): p. 142-148.

65. Hilborn, R., et al., Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences of the United States of America, 2003. 100(11): p. 6564-6568.

66. Lindley, S.T., et al., What caused the Sacramento River fall Chinook stock collapse? 2009, Pacific Fishery Management Council.

67. Hilborn, R., et al., State of the world's fisheries. Annual Review of Environment and Resources, 2003. 28: p. 359-399.

68. Puget Sound Technical Recovery Team, Planning ranges and preliminary guidelines for the delisting and recovery of the Puget Sound Chinook salmon evolutionarily significant unit. 2002.

69. Pimm, S.L., The complexity and stability of ecosystems. Nature, 1984. 307(5949): p. 321-326.

70. Odum, E.P., Strategy of ecosystem development. Science, 1969. 164(3877): p. 262-270.

71. Odum, E.P., Trends expected in stressed ecosystems. BioScience, 1985. 35(7): p. 419-422.

72. Pacala, S. and A. Kinzig, Introduction to theory and the common ecosystem model, in Functional consequences of biodiversity: empirical progress and theoretical extensions, A. Kinzig, S. Pacala, and D. Tilman, Editors. 2002, Princeton University Press: Princeton, NJ, USA. p. 169ñ174.

73. Srivastava, D.S. and M. Vellend, Biodiversity-ecosystem function research: is it relevant to conservation? Annual Review of Ecology, Evolution, and Systematics, 2005. 36: p. 267-294.

74. Williams, R.J. and N.D. Martinez, Simple rules yield complex food webs. Nature, 2000. 404(6774): p. 180-183.

75. Laliberte, E. and P. Legendre, A distance-based framework for measuring functional diversity from multiple traits. Ecology, 2010. 91(1): p. 299-305.

76. Puget Sound Partnership, Ecosystem Status and Trends. 2009, Puget Sound Partnership.

77. Parrish, J.D., D.P. Braun, and R.S. Unnasch, Are we conserving what we say we are? Measuring ecological integrity within protected areas. Bioscience, 2003. 53(9): p. 851-860.

78. The Nature Conservancy, Advanced Guidance for Step 3: Assessing the Viability of the Focal Conservation Targets. 2007, The Nature Conservancy.

79. Fancy, S.G., J.E. Gross, and S.L. Carter, Monitoring the condition of natural resources in US national parks. Environmental Monitoring and Assessment, 2009. 151(1-4): p. 161-174.

80. Poff, N.L., et al., The natural flow regime. Bioscience, 1997. 47(11): p. 769-784.

81. Poff, N.L. and J.K.H. Zimmerman, Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology, 2010. 55(1): p. 194-205.

82. Fu, G.B., M.E. Barber, and S.L. Chen, Hydro-climatic variability and trends in Washington State for the last 50 years. Hydrological Processes, 2010. 24(7): p. 866-878.

83. Barnett, T.P., et al., Human-induced changes in the hydrology of the western United States. Science, 2008. 319(5866): p. 1080-1083.

84. Arhonditsis, G.B. and M.T. Brett, Eutrophication model for Lake Washington (USA) Part I. Model description and sensitivity analysis. Ecological Modelling, 2005. 187(2-3): p. 140-178.

85. Hamilton, D.P. and S.G. Schladow, Prediction of water quality in lakes and reservoirs .1. Model description. Ecological Modelling, 1997. 96(1-3): p. 91-110.

86. Mackas, D.L. and P.J. Harrison, Nitrogenous nutrient sources and sinks in the Juan de Fuca Strait Strait of Georgia Puget Sound estuarine system: Assessing the potential for eutrophication. Estuarine Coastal and Shelf Science, 1997. 44(1): p. 1-21.

87. Connolly, T.P., et al., Processes influencing seasonal hypoxia in the northern California Current System. Journal of Geophysical Research-Oceans, 2010. 115.

88. Leonov, D. and M. Kawase, Sill dynamics and fjord deep water renewal: Idealized modeling study. Continental Shelf Research, 2009. 29(1): p. 221-233.

89. Moore, S.K., et al., Local and large-scale climate forcing of Puget Sound oceanographic properties on seasonal to interdecadal timescales. Limnology and Oceanography, 2008. 53(5): p. 1746-1758.

90. Moore, S.K., et al., A descriptive analysis of temporal and spatial patterns of variability in Puget Sound oceanographic properties. Estuarine Coastal and Shelf Science, 2008. 80(4): p. 545-554.

91. Paulson, A.J., H.C. Curl, and R.A. Feely, The Biogeochemistry of Nutrients and Trace-Metals In Hood Canal, A Puget-Sound Fjord. Marine Chemistry, 1993. 43(1-4): p. 157-173.

92. Washington State Department of Ecology, Evaluating Criteria for the Protection of Freshwater Aquatic Life in Washington's Surface Water Quality Standards: Dissolved Oxygen. Draft Discussion Paper and Literature Summary. 2002: Olympia, WA. p. 90.

93. Washington State Department of Ecology, Evaluating Standards for Protecting Aquatic Life in Washington's Surface Water Quality Standards. Temperature Criteria. Draft Discussion Paper and Literature Summary. 2002, Washington State Department of Ecology: Olympia, WA.

94. Puget Sound Partnership, Puget Sound Action Agenda. 2009, Puget Sound Partnership: Olympia, WA.

95. Bernhard, A.E. and E.R. Peele, Nitrogen limitation of phytoplankton in a shallow embayment in northern Puget Sound. Estuaries, 1997. 20(4): p. 759-769.

96. Carlson, R.E., TROPHIC STATE INDEX FOR LAKES. Limnology and Oceanography, 1977. 22(2): p. 361-369.

97. Edmondson, W.T., PHOSPHORUS, NITROGEN, AND ALGAE IN LAKE WASHINGTON AFTER DIVERSION OF SEWAGE. Science, 1970. 169(3946): p. 690-&.

98. Edmondson, W.T., Sixty Years of Lake Washington: a Curriculum Vitae. Lake and Reservoir Management, 1994. 10(2): p. 75 - 84.

99. Edmondson, W.T. and J.T. Lehman, THE EFFECT OF CHANGES IN THE NUTRIENT INCOME ON THE CONDITION OF LAKE WASHINGTON. Limnology and Oceanography, 1981. 26(1): p. 1-29.

100. Howarth, R.W. and R. Marino, Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. Limnology and Oceanography, 2006. 51(1): p. 364-376.

101. Smith, V.H., Eutrophication of freshwater and coastal marine ecosystems - A global problem. Environmental Science and Pollution Research, 2003. 10(2): p. 126-139.

102. Anderson, D.M., et al., Harmful algal blooms and eutrophication: Examining linkages from selected coastal regions of the United States. Harmful Algae, 2008. 8(1): p. 39-53.

103. Orr, J.C., et al., Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature, 2005. 437(7059): p. 681-686.

104. Federal Water Pollution Control Act, in Title 33, Chapter 26, Subchapter § 1251. As Amended 2002: United States.

105. Walmsley, J.J., Framework for measuring sustainable development in catchment systems. Environmental Management, 2002. 29(2): p. 195-206.

106. DeGasperi, C.L., et al., Linking Hydrological Alteration to Biological Impairment in Urbanizing Streams of the Puget Lowland, Washington, USA. Journal of the American Water Resources Association, 2009. 45(2): p. 512-533.

107. Puget Sound Partnership, Ecosystem Status & Trends, in A 2009 Supplement to State of the Sound Reporting. 2009.

108. Olden, J.D. and N.L. Poff, Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications, 2003. 19(2): p. 101-121.

109. Doren, R.F., et al., Ecological indicators for system-wide assessment of the greater everglades ecosystem restoration program. Ecological Indicators, 2009. 9: p. S2-S16.

110. O'Connor, J.S. and R.T. Dewling, Indices of marine degradation: their utility. Environmental Management, 1986. 10(3): p. 335-343.

111. Noss, R.F., Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology, 1990. 4(4): p. 355-364.

112. Jackson, L.E., J. Kurtz, and W.S. Fisher, Evaluation guidelines for ecological indicators. EPA/620/R-99/005. U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC. 107pp., 2000.

113. Rice, J., Environmental health indicators. Ocean & Coastal Management, 2003. 46(3-4): p. 235-259.

114. Jorgensen, S.E., R. Costanza, and F.L. Xu, Handbook of ecological indicators for assessment of ecosystem health. 2005, Boca Raton, FL, USA: CRC Press.

115. Jennings, S., Indicators to support an ecosystem approach to fisheries. Fish and Fisheries, 2005. 6(3): p. 212-232.

116. Puget Sound Partnership, 2009 State of the Sound Report. 2010: Olympia.

117. Rothaus, D.P., B. Vadopalas, and C.S. Friedman, Precipitous declines in pinto abalone (Haliotis kamtschatkana kamtschatkana) abundance in the San Juan Archipelago, Washington, USA, despite statewide fishery closure. Canadian Journal of Fisheries and Aquatic Sciences, 2008. 65(12): p. 2703(9).

118. Samhouri, J.F., P.S. Levin, and C.J. Harvey, Quantitative Evaluation of Marine Ecosystem Indicator Performance Using Food Web Models. Ecosystems, 2009. 12(8): p. 1283-1298.

119. Slater, L. and G.V. Byrd, Status, trends, and patterns of covariation of breeding seabirds at St Lazaria Island, Southeast Alaska, 1994-2006. Journal of Biogeography, 2009. 36(Special Issue): p. 465-475.

120. Baldwin, D.H., et al., A fish of many scales: extrapolating sublethal pesticide exposures to the productivity of wild salmon populations. Ecological Applications, 2009. 19(8): p. 2004-2015.

121. Fahrig, L. and G. Merriam, Conservation of fragmented populations. Conservation Biology, 1994. 8(1): p. 50-59.

122. Krummel, J.R., et al., Landscape patterns in a disturbed environment. Oikos, 1987. 48: p. 321-324.

123. Senft, R.L., et al., Large herbivore foraging and ecological hierarchies. Bioscience, 1987. 37: p. 789-799.

124. Simenstad, C.A., et al., Historic Change of Puget Sound Shorelines: Puget Sound Nearshore Ecosystem Project Change Analysis. Puget Sound Nearshore Report No. 2010-XX. 2010 (in review), Published by Washington Department of Fish and Wildlife, Olympia, Washington, and U.S. Army Corps of Engineers: Seattle, Washington.

125. Puget Sound Partnership, Water Quantity Topic Forum. 2008, Puget Sound Partnership.

126. Newton, J.A., et al., Washington State Marine Water Quality, 1998 through 2000. 2002, Washington State Department of Ecology, Environmental Assessment Program: Olympia, WA.

127. Babson, A.L., A. Kawase, and P. MacCready, Seasonal and interannual variability in the circulation of Puget Sound, Washington: A box model study. Atmosphere-Ocean, 2006. 44(1): p. 29-45.

128. Cokelet, E.D., R.J. Stewart, and C.C. Ebbesmeyer. The Annual Mean Transport and Refluxing in Puget Sound. in First Annual Meeting on Puget Sound Research. 1991. Seattle, WA: Puget Sound Water Quality Authority.

129. Li, M., A. Gargett, and K. Denman, Seasonal and internannual variability of estuarine circulation in a box model of the Strait of Georgia and Juan de Fuca Strait. Atmosphere-Ocean, 1999. 37(1): p. 1-19.

130. Washington Administrative Code (WAC), Surface Water Criteria - Freshwater, in WAC 173-201A-200, Washington State Department of Ecology.

131. Washington Administrative Code (WAC), Surface Water Criteria - Marine Water in WAC 173-201A-210, Washington State Department of Ecology.

132. Cloern, J.E., Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology-Progress Series, 2001. 210: p. 223-253.

133. Kemp, W.M., et al., Eutrophication of Chesapeake Bay: historical trends and ecological interactions. Marine Ecology-Progress Series, 2005. 303: p. 1-29.

134. Ryther, J.H. and W.M. Dunstan, NITROGEN,PHOSPHORUS, AND EUTROPHICATION IN COASTAL MARINE ENVIRONMENT. Science, 1971. 171(3975): p. 1008-&.

135. Schindler, D.W., Recent advances in the understanding and management of eutrophication. Limnology and Oceanography, 2006. 51(1): p. 356-363.

136. Moore, A. and M. Hicks, Nutrient Criteria Development in Washington State - Phosphorus. 2004, Washington State Department of Ecology: Olympia, WA.

137. Hallock, D., River and Stream Ambient Water Quality Monitoring Report: Water Year 2009. 2009, Washington State Department of Ecology: Olympia, WA.

138. Washington Administrative Code (WAC), Surface Water Criteria - Use designations - fresh waters, in WAC 173-201A-600, Washington State Department of Ecology.

139. Cude, C.G., Reply to discussion by David G. Smith, Robert J. Davies-Colley, and John W. Nagels - "Oregon water quality index: A tool for evaluating water quality management". Journal of the American Water Resources Association, 2002. 38(1): p. 315-318.

140. Cude, C.G., Oregon Water Quality Index: A tool for evaluating water quality management effectiveness. Journal of the American Water Resources Association, 2001. 37(1): p. 125-137.

141. Smith, D.G., R.J. Davies-Colley, and J.W. Nagels, Discussion - "Oregon Water Quality Index: A tool for evaluating water quality management," by Curtis G. Cude. Journal of the American Water Resources Association, 2002. 38(1): p. 313-314.

142. Kaurish, F.W. and T. Younos, Developing a standardized water quality index for evaluating surface water quality. Journal of the American Water Resources Association, 2007. 43(2): p. 533-545.

143. Canadian Council of Ministers of the Environment, Canadian water quality guidelines for the protection of aquatic life: CCME Water Quality Index 1.0, User's Manual, in Canadian environmental quality guidelines, 1999. 2001, Canadian Council of Ministers of the Environment: Winnipeg.

144. EVS Environmental Consultants, Status, Trends and Effects of Toxic Contaminants in the Puget Sound Environment. 2003, Puget Sound Action Team.

145. Hart Crowser Inc., et al., Phase 1: Initial Estimate of Toxic Chemical Loadings to Puget Sound. 2007, Washington State Department of Ecology: Olympia, WA.

146. Puget Sound Action Team, Toxics in Puget Sound. 2006, Puget Sound Action Team: Olympia, WA.

147. Puget Sound Partnership, Water Quality Topic Forum. 2008, Puget Sound Partnership.

148. Schwarzenbach, R.P., et al., The challenge of micropollutants in aquatic systems. Science, 2006. 313(5790): p. 1072-1077.

149. Sumpter, J.P., Protecting aquatic organisms from chemicals: the harsh realities. Philosophical Transactions of the Royal Society a-Mathematical Physical and Engineering Sciences, 2009. 367(1904): p. 3877-3894.

150. Kolpin, D.W., et al., Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999-2000: A national reconnaissance. Environmental Science & Technology, 2002. 36(6): p. 1202-1211.

151. Barnes, K.K., et al., A national reconnaissance of pharmaceuticals and other organic wastewater contaminants in the United States -- I) Groundwater. Science of The Total Environment, 2008. 402(2-3): p. 192-200.

152. Focazio, M.J., et al., A national reconnaissance for pharmaceuticals and other organic wastewater contaminants in the United States -- II) Untreated drinking water sources. Science of The Total Environment, 2008. 402(2-3): p. 201-216.

153. Loos, R., et al., EU-wide survey of polar organic persistent pollutants in European river waters. Environmental Pollution, 2009. 157(2): p. 561-568.

154. Reemtsma, T., et al., Polar pollutants entry into the water cycle by municipal wastewater: A European perspective. Environmental Science & Technology, 2006. 40(17): p. 5451-5458.

155. King County, Survey of Endocrine Disruptors in King County Surface Waters, R. Jack and D. Lester, Editors. 2007, King County, Water and Land Resources Divisiuon: Seattle, WA.

156. Pomati, F., et al., Effects and interactions in an environmentally relevant mixture of pharmaceuticals. Toxicological Sciences, 2008. 102(1): p. 129-137.

157. Brian, J.V., et al., Accurate prediction of the response of freshwater fish to a mixture of estrogenic chemicals. Environmental Health Perspectives, 2005. 113(6): p. 721-728.

158. Filby, A.L., et al., Health Impacts of Estrogens in the Environment, Considering Complex Mixture Effects. Environ Health Perspect, 2007. 115(12).

159. Sumpter, J.P. and A.C. Johnson, Lessons from endocrine disruption and their application to other issues concerning trace organics in the aquatic environment. Environmental Science & Technology, 2005. 39(12): p. 4321-4332.

160. Myers, M.S., et al., Toxicopathic hepatic lesions as biomarkers of chemical contaminant exposure and effects in marine bottomfish species from the Northeast and Pacific Coasts, USA. Marine Pollution Bulletin, 1998. 37(1-2): p. 92-113.

161. Myers, M.S., L.L. Johnson, and T.K. Collier, Establishing the causal relationship between polycyclic aromatic hydrocarbon (PAH) exposure and hepatic neoplasms and neoplasia-related liver lesions in English sole (Pleuronectes vetulus). Human and Ecological Risk Assessment, 2003. 9(1): p. 67-94.

162. Myers, M.S., et al., Improved flatfish health following remediation of a PAH-contaminated site in Eagle Harbor, Washington. Aquatic Toxicology, 2008. 88(4): p. 277-288.

163. Stehr, C.M., et al., Toxicopathic liver lesions in English sole and chemical contaminant exposure in Vancouver Harbour, Canada. Marine Environmental Research, 2004. 57(1-2): p. 55-74.

164. Johnson, L.L., T.K. Collier, and J.E. Stein, An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conservation-Marine and Freshwater Ecosystems, 2002. 12(5): p. 517-538.

165. Carls, M.G. and J.P. Meador, A Perspective on the Toxicity of Petrogenic PAHs to Developing Fish Embryos Related to Environmental Chemistry. Human and Ecological Risk Assessment, 2009. 15(6): p. 1084-1098.

166. Sumpter, J.P. and S. Jobling, VITELLOGENESIS AS A BIOMARKER FOR ESTROGENIC CONTAMINATION OF THE AQUATIC ENVIRONMENT. Environmental Health Perspectives, 1995. 103: p. 173-178.

167. Johnson, L.L., et al., Xenoestrogen exposure and effects in English sole (Parophrys vetulus) from Puget Sound, WA. Aquatic Toxicology, 2008. 88(1): p. 29-38.

168. Kortner, T.M., S.H. Vang, and A. Arukwe, Modulation of salmon ovarian steroidogenesis and growth factor responses by the xenoestrogen, 4-nonylphenol. Chemosphere, 2009. 77(7): p. 989-998.

169. Lyssimachou, A. and A. Arukwe, Alteration of brain and interrenal StAR protein, P450scc, and Cyp11 beta mRNA levels in Atlantic salmon after nominal waterborne exposure to the synthetic pharmaceutical estrogen ethynylestradiol. Journal of Toxicology and Environmental Health-Part a-Current Issues, 2007. 70(7): p. 606-613.

170. Arukwe, A., Steroidogenic acute regulatory (StAR) protein and cholesterol side-chain cleavage (P450scc)-regulated steroidogenesis as an organ-specific molecular and cellular target for endocrine disrupting chemicals in fish. Cell Biology and Toxicology, 2008. 24(6): p. 527-540.

171. Collier, T.K., et al., A comprehensive assessment of the impacts of contaminants on fish from an urban waterway. Marine Environmental Research, 1998. 46(1-5): p. 243-247.

172. Wang, X.H., et al., Polycyclic aromatic hydrocarbon (PAH) metabolites in marine fishes as a specific biomarker to indicate PAH pollution in the marine coastal environment. Journal of Environmental Science and Health Part a-Toxic/Hazardous Substances & Environmental Engineering, 2008. 43(3): p. 219-226.

173. Stein, J.E., et al., CONTAMINANT EXPOSURE AND BIOCHEMICAL EFFECTS IN OUTMIGRANT JUVENILE CHINOOK SALMON FROM URBAN AND NONURBAN ESTUARIES OF PUGET-SOUND, WASHINGTON. Environmental Toxicology and Chemistry, 1995. 14(6): p. 1019-1029.

174. Johnson, L.L., et al., Contaminant exposure in outmigrant juvenile salmon from Pacific Northwest estuaries of the United States. Environmental Monitoring and Assessment, 2007. 124(1-3): p. 167-194.

175. McCain, B.B., et al., UPTAKE OF AROMATIC AND CHLORINATED HYDROCARBONS BY JUVENILE CHINOOK SALMON (ONCORHYNCHUS-TSHAWYTSCHA) IN AN URBAN ESTUARY. Archives of Environmental Contamination and Toxicology, 1990. 19(1): p. 10-16.

176. Meador, J.P., T.K. Collier, and J.E. Stein, Determination of a tissue and sediment threshold for tributyltin to protect prey species of juvenile salmonids listed under the US Endangered Species Act. Aquatic Conservation-Marine and Freshwater Ecosystems, 2002. 12(5): p. 539-551.

177. Meador, J.P., T.K. Collier, and J.E. Stein, Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juvenile salmonids listed under the US Endangered Species Act. Aquatic Conservation-Marine and Freshwater Ecosystems, 2002. 12(5): p. 493-516.

178. Sloan, C.A., et al., Polybrominated Diphenyl Ethers in Outmigrant Juvenile Chinook Salmon from the Lower Columbia River and Estuary and Puget Sound, Washington. Archives of Environmental Contamination and Toxicology, 2010. 58(2): p. 403-414.

179. Johnson, L.L., et al., Contaminant effects on ovarian development and spawning success in rock sole from Puget Sound, Washington. Transactions of the American Fisheries Society, 1998. 127(3): p. 375-392.

180. O'Connor, T.P. and G.G. Lauenstein, Trends in chemical concentrations in mussels and oysters collected along the US coast: Update to 2003. Marine Environmental Research, 2006. 62(4): p. 261-285.

181. O'Connor, T.P., National distribution of chemical concentrations in mussels and oysters in the USA. Marine Environmental Research, 2002. 53(2): p. 117-143.

182. Baumard, P., et al., Concentrations of PAHs (polycyclic aromatic hydrocarbons) in various marine organisms in relation to those in sediments and to trophic level. Marine Pollution Bulletin, 1998. 36(12): p. 951-960.

183. Krishnakumar, P.K., E. Casillas, and U. Varanasi, EFFECT OF ENVIRONMENTAL CONTAMINANTS ON THE HEALTH OF MYTILUS-EDULIS FROM PUGET-SOUND, WASHINGTON, USA .1. CYTOCHEMICAL MEASURES OF LYSOSOMAL RESPONSES IN THE DIGESTIVE CELLS USING AUTOMATIC IMAGE-ANALYSIS. Marine Ecology-Progress Series, 1994. 106(3): p. 249-261.

184. Krishnakumar, P.K., et al., Environmental contaminants and the prevalence of hemic neoplasia (leukemia) in the common mussel (Mytilus edulis complex) from Puget sound, Washington, USA. Journal of Invertebrate Pathology, 1999. 73(2): p. 135-146.

185. West, J.E., S.M. O'Neill, and G.M. Ylitalo, Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (Clupea pallasi) populations in the Puget Sound (USA) and Strait of Georgia (Canada). Science of The Total Environment, 2008. 394(2-3): p. 369-378.

186. O'Neill, S.M. and J.E. West, Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society, 2009. 138(3): p. 616-632.

187. Quinn, T.P., The behavior and ecology of Pacific salmon and trout, ed. T.P. Quinn. 2005, Seattle, WA: University of Washington Press.

188. Cullon, D.L., et al., PERSISTENT ORGANIC POLLUTANTS IN CHINOOK SALMON (ONCORHYNCHUS TSHAWYTSCHA): IMPLICATIONS FOR RESIDENT KILLER WHALES OF BRITISH COLUMBIA AND ADJACENT WATERS. Environmental Toxicology and Chemistry, 2009. 28(1): p. 148-161.

189. Meador, J.P., et al., Bioaccumulation of polychlorinated biphenyls in juvenile chinook salmon (Oncorhynchus tshawytscha) outmigrating through a contaminated urban estuary: dynamics and application. Ecotoxicology, 2010. 19(1): p. 141-152.

190. Ross, P.S., et al., Harbor seals (Phoca vitulina) in British Columbia, Canada, and Washington State, USA, reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry, 2004. 23(1): p. 157-165.

191. Ross, P.S., Fireproof killer whales (Orcinus orca): flame-retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Sciences, 2006. 63(1): p. 224-234.

192. Cayan, D.R., Interannual climate variability and snowpack in the western United States. Journal of Climate, 1996. 9(5): p. 928-948.

193. Bach, A., Snowshed contributions to the Nooksack river watershed, North Cascades Range, Washington. Geographical Review, 2002. 92(2): p. 192-212.

194. Mote, P., et al., Impacts of climate variability and change inthe Pacific Northwest. 1999, The JISAO Climate Impacts Group, University of Washington: Seattle, WA.

195. Hamlet, A.F., et al., Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate, 2005. 18(21): p. 4545-4561.

196. Mote, P., A. Hamlet, and E. Salathe, Has spring snowpack declined in the Washington Cascades? Hydrology and Earth System Sciences, 2008. 12(1): p. 193-206.

197. Stewart, I.T., D.R. Cayan, and M.D. Dettinger, Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change, 2004. 62(1-3): p. 217-232.

198. Stewart, I.T., D.R. Cayan, and M.D. Dettinger, Changes toward earlier streamflow timing across western North America. Journal of Climate, 2005. 18(8): p. 1136-1155.

199. Luce, C.H. and Z.A. Holden, Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters, 2009. 36.

200. Elsner, M.M., et al., Implications of 21st Century Climate Change for the Hydrology of Washington State, in Washington Climate Change Impacts Assessment. 2009, JISAO Climate Impacts Group, University of Washington: Seattle, WA.

201. Washington State County Growth Management Population Projections: 2000 to 2030. 2007; Available from: http://www.ofm.wa.gov/pop/gma/projections07.asp.

202. Simmons, D.L. and R.J. Reynolds, EFFECTS OF URBANIZATION ON BASE-FLOW OF SELECTED SOUTH-SHORE STREAMS, LONG-ISLAND, NEW-YORK. Water Resources Bulletin, 1982. 18(5): p. 797-805.

203. Stover, S.C. and D.R. Montgomery, Channel change and flooding, Skokomish River, Washington. Journal of Hydrology, 2001. 243(3-4): p. 272-286.

204. Burges, S.J., M.S. Wigmosta, and J.M. Meena, Hydrological Effects of Land-Use Change in a Zero-Order Catchment. Journal of Hydrologic Engineering, 1998. 3(2): p. 86-97.

205. Leith, R.M. and P.H. Whitfield, Some effects of urbanization on streamflow records in a small watershed in the lower Fraser Valley, BC. Northwest Science, 2000. 74(1): p. 69-75.

206. Moscrip, A.L. and D.R. Montgomery, Urbanization, flood frequency, and salmon abundance in Puget Lowland streams. Journal of the American Water Resources Association, 1997. 33(6): p. 1289-1297.

207. Konrad, C.P. and D. Booth, Hydrologic Trends Associated with Urban Development for Selected Streams in the Puget Sound Basin, Western Washington. 2002, U.S. Geological Survey: Tacoma, WA.

208. Fleming, S.W., Quantifying urbanization-associated changes in terrestrial hydrologic system memory. Acta Geophysica, 2007. 55(3): p. 359-368.

209. Scott, J.B., C.R. Steward, and Q.J. Stober, EFFECTS OF URBAN-DEVELOPMENT ON FISH POPULATION-DYNAMICS IN KELSEY CREEK, WASHINGTON. Transactions of the American Fisheries Society, 1986. 115(4): p. 555-567.

210. Pess, G.R., et al., Landscape characteristics, land use, and coho salmon (Oncorhynchus kisutch) abundance, Snohomish River, Wash., USA. Canadian Journal of Fisheries and Aquatic Sciences, 2002. 59(4): p. 613-623.

211. Bilby, R.E. and L.A. Mollot, Effect of changing land use patterns on the distribution of coho salmon (Oncorhynchus kisutch) in the Puget Sound region. Canadian Journal of Fisheries and Aquatic Sciences, 2008. 65(10): p. 2138-2148.

212. Paul, M.J. and J.L. Meyer, Streams in the urban landscape. Annual Review of Ecology and Systematics, 2001. 32: p. 333-365.

213. Montgomery, D.R., et al., Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. Canadian Journal of Fisheries and Aquatic Sciences, 1996. 53(5): p. 1061-1070.

214. Schuett-Hames, D.E., et al., Patterns of gravel scour and fill after spawning by chum salmon in a western Washington stream. North American Journal of Fisheries Management, 2000. 20(3): p. 610-617.

215. Beamish, R.J., et al., A RELATIONSHIP BETWEEN FRASER-RIVER DISCHARGE AND INTERANNUAL PRODUCTION OF PACIFIC SALMON (ONCORHYNCHUS SPP) AND PACIFIC HERRING (CLUPEA-PALLASI) IN THE STRAIT OF GEORGIA. Canadian Journal of Fisheries and Aquatic Sciences, 1994. 51(12): p. 2843-2855.

216. Battin, J., et al., Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America, 2007. 104(16): p. 6720-6725.

217. Mathews, S.B. and F.W. Olson, FACTORS AFFECTING PUGET SOUND COHO SALMON (ONCORHYNCHUS-KISUTCH) RUNS. Canadian Journal of Fisheries and Aquatic Sciences, 1980. 37(9): p. 1373-1378.

218. Bauer, S.B. and S.C. Ralph, Strengthening the use of aquatic habitat indicators in Clean Water Act programs. Fisheries, 2001. 26(6): p. 14-+.

219. Matzen, D.A. and H.B. Berge, Assessing small-stream biotic integrity using fish assemblages across an urban landscape in the puget sound lowlands of western Washington. Transactions of the American Fisheries Society, 2008. 137(3): p. 677-689.

220. Morley, S.A. and J.R. Karr, Assessing and restoring the health of urban streams in the Puget Sound basin. Conservation Biology, 2002. 16(6): p. 1498-1509.

221. Bond, N.R. and B.J. Downes, The independent and interactive effects of fine sediment and flow on benthic invertebrate communities characteristic of small upland streams. Freshwater Biology, 2003. 48(3): p. 455-465.

222. Cassin, J.L., et al., Development of Hydrological and Biological Indicators of Flow Alteration in Puget Sound Lowland Streams. 2005, King County Water and Land Resources Division: Seattle, Washington.

223. Recchia, C. and E. Whiteman, Draft North Central Coast Marine Protected Areas Monitoring Plan. 2009, California Ocean Science Trust: Oakland, CA.

224. McLeod, K.L., et al., Scientific consensus statement on marine ecosystem-based management. Signed by 221 academic scientists and policy experts with relevant expertise and published by the Communication Partnership for Science and the Sea at http://compassonline.org/?q=EBM, 2005.

225. World Health Organization, Constitution of the World Health Organization. Official Records World Health Organization, 1948. 2: p. 100.

226. Purcell, J.E., S. Uye, and W.T. Lo, Anthropogenic causes of jellyfish blooms and their direct consequences for humans: a review. Marine Ecology Progress Series, 2007. 350: p. 153-174.

227. Heithaus, M.R., et al., Predicting ecological consequences of marine top predator declines. Trends in Ecology & Evolution, 2008. 23(4): p. 202-210.

228. Hagenaars, A. and B. Praag, A synthesis of poverty line definitions. Review of Income and Wealth, 1985. 31(2): p. 139-154.

229. Gerber, L.R. and L.T. Hatch, Are we recovering? An evaluation of recovery criteria under the U.S. Endangered Species Act. Ecological Applications, 2002. 12(3): p. 668-673.

230. Walters, C.J. and S.J.D. Martell, Fisheries ecology and management. 2004, Princeton, NJ, USA: Princeton University Press.

231. Schindler, D.W., Eutrophication and recovery in experimental lakes: implications for lake management. Science, 1974. 184(4139): p. 897-899.

232. Suter, G.W., Ecological risk assessment. 2007, Boca Raton, FL: CRC Press.

233. Tallis, H.M., et al., The many faces of ecosystem-based management: making the process work today in real places. Marine Policy, 2010. 34(2): p. 340-348.

234. Cheung, W.W.L. and U.R. Sumaila, Trade-offs between conservation and socio-economic objectives in managing a tropical marine ecosystem. Ecological Economics, 2008. 66(1): p. 193-210.

235. Selecky, M.C., J. Hardy, and G. Palcisko, Human Health Evaluation of Contaminants in Puget Sound Fish. 2006, Washington State Department of Health: Olympia, WA.

236. Jennings, S. and N.K. Dulvy, Reference points and reference directions for size-based indicators of community structure. ICES J. Mar. Sci., 2005. 62(3): p. 397-404.

237. Failing, L. and R. Gregory, Ten common mistakes in designing biodiversity indicators for forest policy. Journal of Environmental Management, 2003. 68(2): p. 121-132.

238. Donaldson, R. and H. Kim, Price barriers in the Dow Jones industrial average. Journal of Financial and Quantitative Analysis, 1993. 28(3): p. 313-330.

239. National Bureau of Economic Research, Determination of the December 2007 Peak in Economic Activity. 2008, National Bureau of Economic Research. 7pp.

240. Elmer-DeWitt, P. Tech sector snaphot: Apple's shifting fortunes. http://brainstormtech.blogs.fortune.cnn.com/tag/sector-snapshot/ 2008; Available from: http://brainstormtech.blogs.fortune.cnn.com/tag/sector-snapshot/.

241. DeLuca, W.V., et al., Coastal urbanization and the integrity of estuarine waterbird communities: threshold responses and the importance of scale. Biological Conservation, 2008. 141(11): p. 2669-2678.

242. Parrish, J.K., et al., Beached birds and physical forcing in the California Current System. Marine Ecology Progess Series, 2007. 352: p. 275-288.

243. Walpole, M., et al., Tracking progress toward the 2010 biodiversity target and beyond. Science, 2009. 325(5947): p. 1503-1504.

244. Puget Sound Action Team, State of the Sound. 2007, Puget Sound Action Team: Seattle, WA.

245. Caddy, J.F., Limit reference points, traffic lights, and holistic approaches to fisheries management with minimal stock assessment input. Fisheries Research, 2002. 56(2): p. 133-137.

246. Link, J.S., Translating ecosystem indicators into decision criteria. ICES Journal of Marine Science: Journal du Conseil, 2005. 62(3): p. 569-576.

247. Bottrill, M.C., et al., Is conservation triage just smart decision making? Trends in Ecology & Evolution, 2008. 23(12): p. 649-654.

248. The Business Conference Board, Business Cycle Indicators Handbook. 2001, The Business Conference Board. 156pp.

249. National Marine Fisheries Service, Recovery Plan for Southern Resident Killer Whales (Orcinus orca), in National Marine Fisheries Service, Northwest Region, Seattle, WA. 2008.

250. Karr, J.R., Ecological integrity: protecting Earth's life support systems, in Ecosystem health: new goals for environmental management, R. Costanza, B. Norton, and B. Haskell, Editors. 1992, Island Press: Washington, DC. p. 223-238.

251. Pauly, D., Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology & Evolution, 1995. 10(10): p. 430.

252. Castilla, J.C., Coastal marine communities: trends and perspectives from human-exclusion experiments. Trends in Ecology & Evolution, 1999. 14(7): p. 280-283.

253. Dayton, P.K., et al., Marine reserves: Parks, baselines, and fishery enhancement. Bulletin of Marine Science, 2000. 66(3): p. 617-634.

254. Knowlton, N. and J. Jackson, Shifting baselines, local impacts, and global change on coral reefs. Plos Biology, 2008. 6(2): p. e54.

255. Pitcher, T.J., Fisheries managed to rebuild ecosystems? Reconstructing the past to salvage the future. Ecological Applications, 2001. 11(2): p. 601-617.

256. Jackson, J.B.C., et al., Historical overfishing and the recent collapse of coastal ecosystems. Science, 2001. 293: p. 629-638.

257. Aronson, R., et al., Emergent zonation and geographic convergence of coral reefs. Ecology, 2005. 86(10): p. 2586-2600.

258. Butler, V.L. and S.K. Campbell, Resource intensification and resource depression in the Pacific Northwest of North America: A zooarchaeological review. Journal of World Prehistory, 2004. 18(4): p. 327-405.

259. Jackson, J.B., Reefs since Columbus. Coral Reefs, 1997. 16(Supplement): p. S23-S32.

260. Levin, P.S. and A. Dufault, Eating through the food web? An analysis of seafood recipes in Puget Sound. Fish and Fisheries, in press.

261. Lichatowich, J., Salmon without rivers: a history of the Pacific salmon crisis. 2001, Washington, DC: Island Press.

262. Nichols, F., Interdecadal change in the deep Puget Sound benthos. Hydrobiologia, 2003. 493(1): p. 95-114.

263. Lozano-Montes, H.M., T.J. Pitcher, and N. Haggan, Shifting environmental and cognitive baselines in the upper Gulf of California. Frontiers in Ecology and the Environment, 2008. 6(2): p. 75-80.

264. Sáenz-Arroyo, A., et al., Rapidly shifting environmental baselines among fishers of the Gulf of California. Proceedings of the Royal Society B: Biological Sciences, 2005. 272(1575): p. 1957.

265. Ainsworth, C.H., T.J. Pitcher, and C. Rotinsulu, Evidence of fishery depletions and shifting cognitive baselines in Eastern Indonesia. Biological Conservation, 2008. 141(3): p. 848-859.

266. Lester, S.E., et al., Biological effects within no-take marine reserves: a global synthesis. Marine Ecology Progress Series, 2009. 384: p. 33-46.

267. Stewart, G.B., et al., Temperate marine reserves: global ecological effects and guidelines for future networks. Conservation Letters, 2009. 2(6): p. 243-253.

268. Dulvy, N.K., R.P. Freckleton, and N.V. Polunin, Coral reef cascades and the indirect effects of predator removal by exploitation. Ecology Letters, 2004. 7(5): p. 410-416.

269. Friedlander, A.M. and E.E. DeMartini, Contrasts in density, size, and biomass of reef fishes between the northwestern and the main Hawaiian islands: the effects of fishing down apex predators. Marine Ecology Progress Series, 2002. 230: p. 253-264.

270. Sandin, S.A., et al., Baselines and degradation of coral reefs in the northern Line Islands. PLoS ONE, 2008. 3(2): p. e1548.

271. Stallings, C.D., Fishery-independent data reveal negative effect of human population density on Caribbean predatory fish communities. PLoS ONE, 2009. 4(5): p. e5333.

272. Collins, B., D. Montgomery, and A. Haas, Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries and Aquatic Sciences, 2002. 59(1): p. 66-76.

273. McClanahan, T., et al., Toward pristine biomass: Reef fish recovery in coral reef marine protected areas in Kenya. Ecological Applications, 2007. 17(4): p. 1055-1067.

274. Hammond, J., Fish in Puget Sound. The American Angler, 1886. 25(10): p. 392-393.

275. Collins, B. and A. Sheikh, Historical reconstruction, classification, and change analysis of Puget Sound tidal marshes. Final project report to Washington Department of Natural Resources Aquatic Resources Division, Olympia, WA, 2005: p. 98504-7027.

276. Shaw, S.D., et al., Polybrominated Diphenyl Ethers in Marine Ecosystems of the American Continents: Foresight from Current Knowledge. Reviews on Environmental Health. 24: p. 157-229.

277. Palsson, W., R. Pacunski, and T. Parra. Time Will Tell: Long-term Observations of the Response of Rocky-Habitat Fishes to Marine Reserves in Puget Sound. in 2003 Georgia Basin/Puget Sound Research Conference Proceedings. 2004.

278. Tuya, F., M. Soboil, and J. Kido, An assessment of the effectiveness of marine protected areas in the San Juan Islands, Washington, USA. ICES Journal of Marine Science, 2000. 57(4): p. 1218-1226.

279. Thom, R. and L. Hallum, Historical changes in the distribution of tidal marshes, eelgrass meadows and kelp forests in Puget Sound. 1991, Final Report to Office of Puget Sound Region 10, United States Environmental Protection Agency. p. 302-313.

280. Campbell, L., et al., Beyond Baselines: Rethinking Priorities for Ocean Conservation. Ecology and Society, 2009. 14(1): p. 14.

281. Beisner, B.E., D.T. Haydon, and K. Cuddington, Alternative stable states in ecology. Frontiers in Ecology and the Environment, 2003. 1(7): p. 376-382.

282. Scheffer, M., et al., Catastrophic shifts in ecosystems. Nature, 2001. 413: p. 591-596.

283. Muradian, R., Ecological thresholds: a survey. Ecological Economics, 2001. 38(1): p. 7-24.

284. Ling, S.D., et al., Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. Proceedings of the National Academy of Sciences, 2009. 106(52): p. 22341-22345.

285. Steneck, R.S., et al., Kelp forest ecosystems: biodiversity, stability, resilience and future. Environmental Conservation, 2002. 29(04): p. 436-459.

286. Bellwood, D.R., A.S. Hoey, and J.H. Choat, Limited functional redundancy in high diversity systems: resilience and ecosystem function on coral reefs. Ecology Letters, 2003. 6: p. 281-285.

287. Martin, J., et al., Structured decision making as a conceptual framework to identify thresholds for conservation and management. Ecological Applications, 2009. 19(5): p. 1079-1090.

288. Samhouri, J.F., P.S. Levin, and C.H. Ainsworth, Identifying Thresholds for Ecosystem-Based Management. PLoS ONE, 2010. 5(1): p. e8907.

289. Jennings, S., M.J. Kaiser, and J.D. Reynolds, Marine fisheries ecology. 2001, Oxford, UK: Blackwell Publishing.

290. Burnham, K.P. and D.R. Anderson, Model selection and multimodel inference. 2002, New York: Springer.

291. Rice, C., Evaluating the Biological Condition of Puget Sound. 2007, University of Washington, Ph.D. dissertation.

292. Rapport, D., Evaluating ecosystem health. Journal of Aquatic Ecosystem Stress and Recovery (Formerly Journal of Aquatic Ecosystem Health), 1992. 1(1): p. 15-24.

293. Vaske, J., M. Donnelly, and B. Shelby, Establishing management standards: Selected examples of the normative approach. Environmental Management, 1993. 17(5): p. 629-643.

294. Jackson, J., Social Stratification, Social Norms, and Roles, in Current studies in social psychology, I. Steiner and M. Fishbein, Editors. 1965. p. 301-309.

295. Bateman, I., et al., Reducing gain-loss asymmetry: A virtual reality choice experiment valuing land use change. Journal of Environmental Economics and Management, 2009. 58: p. 106-118.

296. McCallum, D. and R. Rollins, Strategies for Use and Protection of the Gulf Islands Marine Environment. Proceedings of the 2005 Puget Sound Georgia Basin Research Conference, 2005.

297. Iceland, C., C. Hanson, and C. Lewis, Identifying important ecosystem goods & services in Puget Sound. 2008, World Resources Institute.

298. Shin, Y.-J., et al., Can simple be useful and reliable? Using ecological indicators to represent and compare the states of marine ecosystems. ICES J. Mar. Sci., 2010: p. fsp287.

299. Briske, D.D., S.D. Fuhlendorf, and F.E. Smeins, A unified framework for assessment and application of ecological thresholds. Rangeland Ecology & Management, 2006. 59(3): p. 225-236.

300. Mengerink, K., A. Schempp, and J. Austin, Ocean and Coastal Ecosystem-Based Management: ImplementatIon Handbook. 2009, Washington, D.C., USA: Enviornmental Law Institute.

301. Washington State Department of Ecololgy. Marine Aquatic Life Uses, http://www.ecy.wa.gov/programs/wq/swqs/reference_files/MarineWQSMap.pdf. 2010; Available from: http://www.ecy.wa.gov/programs/wq/swqs/reference_files/MarineWQSMap.pdf.

302. Code of Federal Regulations (CFR), Title 40: Protection of Environment, Part 131—Water Quality Standards, Subpart D—Federally Promulgated Water Quality Standards, § 131.36

Toxics criteria for those states not complying with Clean Water Act section 303(c)(2)(B). 2006, U.S. Environmental Protection Agency.

303. Stinson, D.W., J. W. Watson, and K. R. McAllister., Washington State status report for the bald eagle. 2007, Washington Dept. of Fish and Wildlife, Wildlife Program: Olympia, WA. p. 86 + viii p.

304. Jeffries, S., et al., Trends and status of harbor seals in Washington state: 1978-1999. Journal of Wildlife Management, 2003. 67(1): p. 207-218.

305. Hayes, G.E. and J.B. Buchanan, Washington State status report for the Peregrine Falcon. Washington Dept. Fish and Wildlife, Olympia. 77 pp., 2002.

306. May, C., Stream-Riparian Ecosystems In the Puget Sound Lowland Eco-Region. A Review of Best Available Science. Watershed Ecology LLC. 74pp., 2003.

307. Mayer, P., et al., Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: A review of current science and regulations. US Environmental Protection Agency, Cincinnati, OH, EPA/600/R-05/118. 40pp., 2005.

308. Washington Administrative Code (WAC), Surface Water Criteria - Establishing lake nutrient criteria in WAC 173-201A-230, Washington State Department of Ecology.

309. Butkus, S., Factors affecting waters with a high pH - Statewide analysis. 2002, Washington State Department of Ecology: Olympia, WA.

310. Washington Administrative Code (WAC), Surface Water Criteria - Toxics Standards and Criteria, in WAC 173-201A-240, Washington State Department of Ecology.

311. Washington Administrative Code (WAC), Instream Resource Protection Program WAC 173-501 through WAC 173-517), Washington State Department of Ecology.

312. Washington State Department of Ecology, A Guide to Instream Flow Setting in Washington State, L.D. Geller, Editor. 2003, Water Resources Program: Olympia, WA.

313. Washington State Department of Ecology, Setting instream flows and allocating water for future out-of-stream uses. 2004: Olympia, WA.

314. United States Geological Survey. WaterWatch - Current water resource conditions. 2010; Available from: http://waterwatch.usgs.gov/?m=real&r=wa&w=real%2Cmap.

315. Washington State Department of Health and Office of Drinking Water. The Minicipal Water law. 2010; Available from: http://www.doh.wa.gov/ehp/dw/municipal_water/mw-law.htm.

CUSPA 2008. Puget Sound UrbanSim Application. Center for Urban Simulation and Policy Analysis, University of Washington. Accessed on 20 April 2010 at http://www.psrc.org/assets/2830/CUSPA_UrbanSim_Doc.pdf

Indicator Evaluation Spreadsheets

1. Species, Food Web and Habitat Indicator Evaluations:

Species, Food Webs and Habitat Spreadsheet

Species, Food Webs and Habitats Literature Cited

Water Quality and Quantity Indicator Evaluations:

Water Quality and Quantity Spreadsheet

Water Quality Literature Cited

Water Quantity Literature Cited

Chapter 1B. Incorporating Human Well-being into Ecosystem-based Management

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[2]Puget Sound Partnership

Section 1. Introduction

The Puget Sound Partnership is charged with identifying actions to protect and restore Puget Sound, and assessing the effectiveness of those actions. As part of its effort to fulfill these charges, the Partnership will identify indicators to monitor the ecological and human systems within the Puget Sound region. These indicators will help inform decision makers and the public about the health of Puget Sound.

In creating the Partnership, the Washington State Legislature identified six goals (State of Washington, 2007):

1. A healthy human population supported by a healthy Puget Sound that is not threatened by changes in the ecosystem;
2. A quality of human life that is sustained by a functioning Puget Sound ecosystem;
3. Healthy and sustaining populations of native species in Puget Sound, including a robust food web;
4. A healthy Puget Sound where freshwater, estuary, nearshore, marine, and upland habitats are protected, restored, and sustained;
5. An ecosystem that is supported by ground water levels as well as river and stream flow levels sufficient to sustain people, fish, and wildlife, and the natural functions of the environment;
6. Fresh and marine waters and sediments of a sufficient quality so that the waters in the region are safe for drinking, swimming, shellfish harvest and consumption, and other human uses and enjoyment, and are not harmful to the native marine mammals, fish, birds, and shellfish of the region.

The first two goals explicitly reference human well-being while the other goals have less direct references or can be indirectly connected to human well-being. Indicators that assess human well-being will therefore be needed to assess the effectiveness of any actions recommended by the Partnership in their Action Agenda (Puget Sound Partnership, 2008).

The use of indicators to track human well-being in previous ecosystem-based management efforts, however, is not common. Indicators connected to human well-being are most often used to measure the effects of social or economic policies and compare these effects across groups. Their use has therefore mostly focused on identifying and using a small set of indicators that covers a particular social or economic system (e.g.., housing or education) affected by the policy. Less common is their use when policy is primarily assessed first in terms of changes in ecological conditions and then only subsequently, if at all, in terms of changes in human conditions.

This report provides a framework for identifying, evaluating, and selecting indicators that track human well-being in the context of ecosystem-based management (EBM). It begins with a discussion of how human well-being can be integrated into EBM and used (in principle) as an over-arching metric by which to evaluate the effectiveness and impacts of management actions. We then give a brief overview of the concept of human well-being, a term that is difficult to define precisely, and discuss the nature of HWB indicators. The following section discusses methods for measuring human well-being and for assessing the links between changes in ecological conditions and changes in human well-being. Finally, the report outlines a framework for cataloging data and empirical studies, and for evaluating the nature and strengths of these links, in a manner that can assist the Puget Sound Partnership in its task of identifying and evaluating potential human well-being and other indicators.

Human Well-being and Ecosystem-based Management

Over the past decade, efforts have been made to expand our understanding of coupled social and ecological systems (Millennium Ecosystem Assessment, 2003; Liu et al., 2007; Walker et al., 2002). Governments at many levels have increasingly sought to base environmental management not just on political considerations, but on goals such as ecological health and resilience. Understanding how the two systems are linked is therefore important. The links between biophysical and human systems, and the support that the biophysical systems provide for human well-being, are both obvious and obviously important. The systematic measurement and assessment of the existence and importance of individual links, however, is less common than simple assertions that such links exist (Bowen and Riley 2003).

Crafting a picture of a linked natural-human system often takes place in the context of ecosystem-based management. In its early conception, EBM was defined to mean "focusing on ecological systems that may cross administrative and political boundaries, incorporating a 'system' perspective sensitive to issues of scale, and managing for ecological integrity" (Endter-Wada, 1998). This initial definition was an ecologically centered view with human systems incorporated simply as political boundaries or more complexly as impacts on the system to be controlled or reduced (Figure 1).

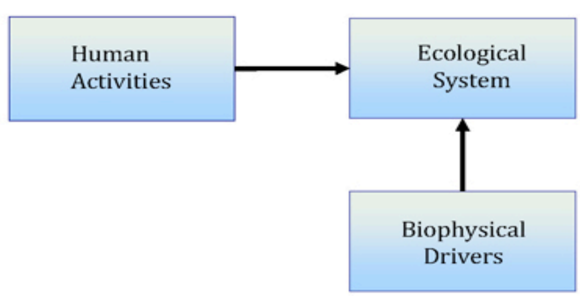

Figure 1: A simplistic view of an ecological system that is affected by but separate from human systems. Adapted from Redman, *et al.* (2004).

Although the purely ecocentric view still exists, there has been increasing recognition of the need to integrate humans and our social systems more completely into the EBM framework. The common approach to EBM has expanded to include the need to manage for the sustainability of human systems as well as ecological communities, to practice adaptive management, and to encourage broad-based involvement and collaboration in implementing EBM. As the term is employed in the Puget Sound region, EBM includes the management of ecosystems in ways that are inclusive of human needs and values, as reflected in the six goals listed in the previous section.

This section provides a conceptual model of how human well-being can be integrated into the Partnership's framework for conducting EBM. This model can also be used to craft a strategy for identifying and evaluating the connections between indicators, biophysical and human-based, and human well-being in the context of the Puget Sound Partnership's tasks. By including human well-being (along with human health) as an explicit goal, the Partnership acknowledges the importance of this integrated view. Including indicators that measure impacts to both the human and biophysical systems will therefore provide stronger support for an EBM effort such as the one being pursued by the Partnership (Bowen and Riley, 2003; Carr et al cite). Bringing HWB into an ecosystem-based management effort has potentially deeper implications, however. The Partnership goals can sometimes conflict with one another, and so the question arises of how to assess and evaluate such conflicts. The current Partnership approach is to compartmentalize the six goals and discuss them separately. Examples of this include the Partnership's Ecosystem Status & Trends document (Puget Sound Partnership, 2009a) and the Identification of Ecosystem Components and Their Indicators and Targets technical memorandum (Puget Sound Partnership, 2009b), where each goal is discussed separately. Connections among the systems represented by the goals are recognized, of course, but the question of how to resolve potential conflicts has not yet been addressed.

Separating ecological goals from human well-being is apparently one way of resolving a long standing tension between adopting a wholly ecocentric or wholly anthropocentric viewpoint in ecosystem-based management (Endter-Wada et al., 1998, for a discussion of this tension). Still, by setting the two sets of goals apart, the Partnership implicitly grants the ecological goals something in the nature of intrinsic value. That is, species, habitat, water quality, and water quantity have value for their own sake; or, it may be that some aspect of a particular goal has value because of its support for aspects of the other natural goals (e.g., the value of nearshore habitat may be derived from its support for certain species), but the goals so supported are still valued for their own sake.

Figure 2 gives a representation of this approach, where actions drawn from the Partnership's Action Agenda can be evaluated in terms of changes to one or more of the Partnership goals (Puget Sound Partnership, 2008). A problem with this construction is the difficulty it creates when intrinsically valued goals conflict with one another or, in this case, with human well-being (Justus et al., 2009). Little guidance is given about which goal should take precedence, and so the resolution of conflicts is hard to assess in a consistent, reasoned way. In contrast, viewing the values involved as instrumental creates an opportunity to evaluate goals with a common metric, because each goal is viewed as an "instrument" in achieving some higher, over-arching goal (Justus et al., 2009).

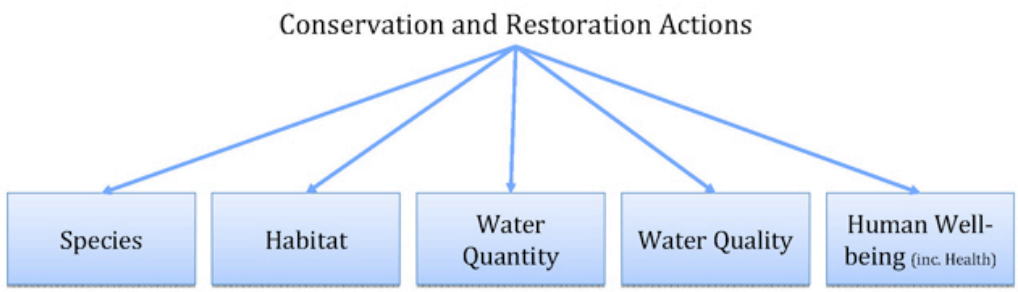

Figure 2: A framework for evaluating the effects of conservation and restoration actions on the Partnership's goals, where each goal is considered with a distinct set of metrics.

In the context of the Puget Sound Partnership, human well-being can be used as such an over-arching goal (Figure 3). Now, the ecological goals are viewed as instrumental in supporting human well-being, which then becomes in principle a common metric by which to assess management actions. "Instrumental" does not mean material or based solely on monetary values. As noted by Justus et al. (2009), something has instrumental value to the extent that it is "considered valuable by valuers" - that is, in the context of EBM, it is something that humans value about the environment. This includes values that are independent of consumption or the use of a resource, for example, and can even involve actions that are to the material detriment of the valuer.

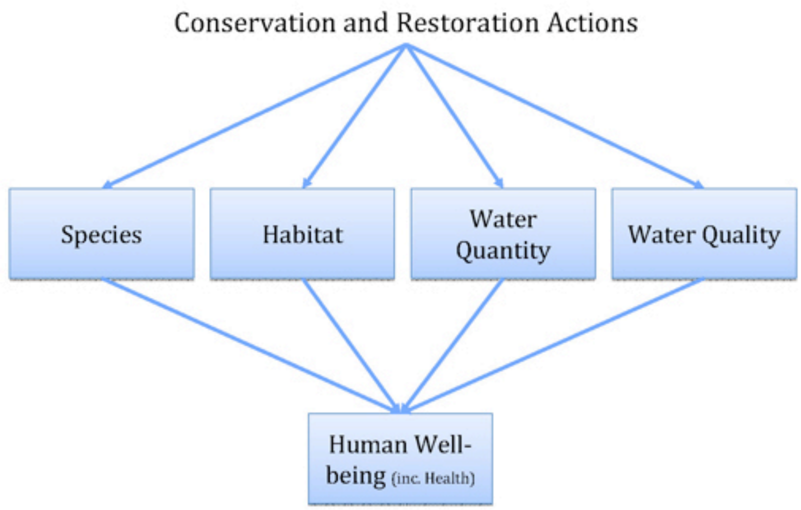

Figure 3: A framework for evaluating the effects of conservation and restoration actions on the Partnership's ecological goals using human well-being as a common metric.

Using this framework, it is straightforward to consider different types of links that connect the ecological goals to HWB, and therefore the different types of instrumental values. In Figure 4, the management objective is to improve the conditions covered by the Species and Water Quantity goals. These goals have direct connections to HWB but through possibly multiple types of values. Figure 5 illustrates a different case, where the ecological goal of Habitat provides indirect value to humans through its ecological connections to the Species and Water Quality goals. Assessing the value in this case would require an understanding of 1) the effect of the action on habitat; 2) the effects of habitat changes on species and water quality; and 3) the value to humans of the resulting changes in the conditions of those two goals.

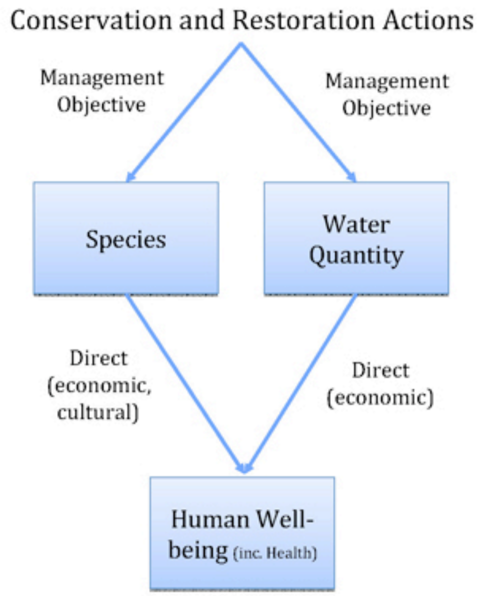

Figure 4: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that are directed at species and water quality.

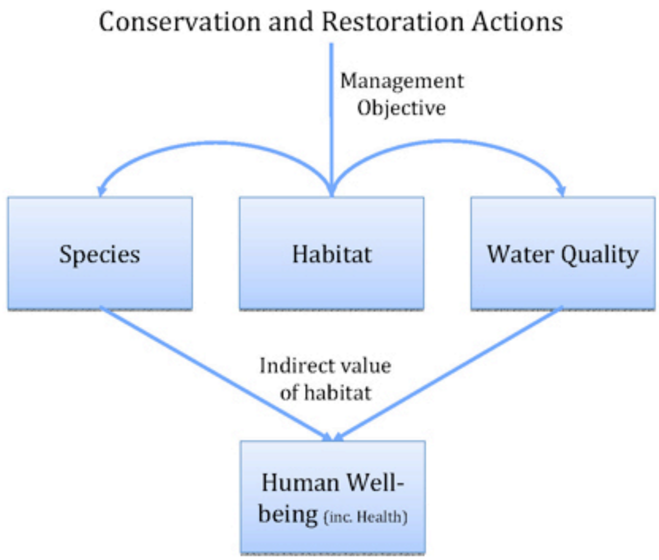

Figure 5: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that are directed at habitat, which in turn affects species and water quality and thereby affecting human well-being.

Creating the links between the Partnership's ecological goals and HWB also points to a more expansive view of the set of relevant indicators. Improving ecological systems is not the only way to improve human well-being (Millennium Ecosystem Assessment, 2005). Many factors support human well-being, only some of which are related to or derived from ecological systems. As Dasgupta (2001) notes, a society's total collection of capital is what supports its well-being. This capital is a diverse collection of traditional forms of capital (buildings and machines), "natural" capital (species and habitats), social capital (examples), and other forms. These forms of capital are partly substitutable for one another, and improvement in human well-being is then possible even if one or two components of total capital decrease (Millennium Ecosystem Assessment, 2005).

Figure 6 shows a simple way of expanding the focus of EBM to encompass other forms of capital that support HWB. In this simple illustration, Economic Activity and Social Conditions are treated as broader social goals because they support human well-being. They are not necessarily objectives for the Partnership's management strategies, however, but are certainly affected by them. Because they have strong links to HWB, assessing the effects on these areas will likely improve management, at least in the case where HWB is used as a common metric. Figure 7 illustrates this by presenting the case where an action improves Habitat by constraining Economic Activity. HWB is enhanced by the first effect through the improvements in the

Species and Water Quality goals, but the constraint on Economic Activity can produce an offsetting negative effect. Accounting for both types of pathways between actions and HWB is necessary to evaluate the total effect of an action.

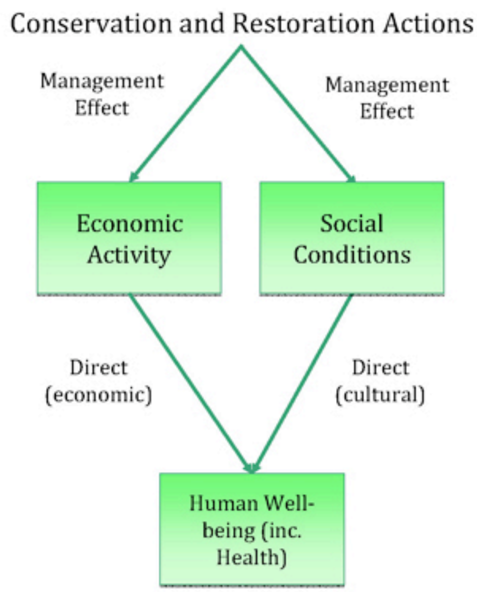

Figure 6: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that affect important components that support human well-being but which are not the objectives of management.

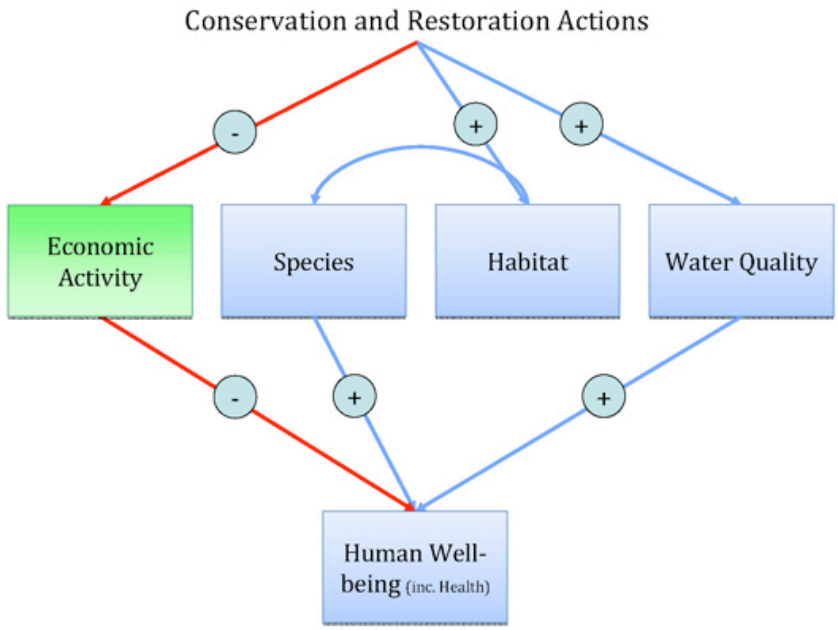

Figure 7: An example of how human well-being can be affected by conservation and restoration actions along multiple pathways.

The framework illustrated in this section can be used to set priorities for actions and help select indicators. The simplicity of the figures, however, masks the incredible number of all the possible pathways that connect HWB to the conservation and restoration actions proposed by the Partnership. The following two sections address this problem. In section 3, we discuss the nature of human well-being and the traditional indicators that have been used to track and register changes in well-being. In section 4, we consider ways in which the various pathways could be evaluated in terms of the "strength" of the connections. Much of that evaluation lies outside the scope of this report, as it involves identifying and evaluating ecological indicators. The discussion in that section considers different approaches for assessing the strength of connections between human well-being and environmental attributes that have direct effects on well-being.

Key Points: Human well-being is both a goal for the Puget Sound Partnership and a potential metric for assessing the effects of conservation and restoration actions that further all Partnership goals.

The Nature of Human Well-being

Human well-being is a broad concept, one that includes many aspects of our everyday lives. It encompasses material well-being, relationships with family and friends, and emotional and physical health. It includes work and recreation, how one feels about one's community, and personal safety. Precisely defining human well-being is difficult, however. Although it can be described, it lacks a universally acceptable definition and has numerous, and often competing, interpretations. As human well-being cannot be directly observed, it cannot be independently measured. And there are a host of terms -- quality of life, welfare, well-living, living standards, utility, life satisfaction, prosperity, needs fulfillment, development, empowerment, capability expansion, human development, poverty, human poverty, land and, more recently, happiness – that are often used interchangeably with human well-being (McGillivray and Clarke, 2008). Despite these difficulties, there is a large body of research covering the subject of human well-being. HWB research occurs in multiple fields such as psychology, medicine, economics, environmental science and sociology (Costanza et al., 2007). In recent times, human well-being has frequently been considered analogous with income and consumption levels. The reasoning goes something like this: humans consume materials and services to meet their needs and desires, and so increase their well-being; markets provide these materials and services; income allows individuals to obtain these market items; therefore, income can be equated with human well-being (Stiglitz et al, 2009). Using income or consumption as a proxy for well-being is problematic, however. Many material goods and services are not marketed; many of the determinants of human well-being are not resources but are circumstances or experiences that still have important connections to human well-being; and even a given market basket can produce varying amounts of HWB depending on the individual, so that some individuals can achieve a higher level of HWB with a market basket (i.e., income) smaller than others. Finally, income measured at the individual or national level overlooks distributional issues that can affect well-being (Stiglitz et al, 2009).

For this exercise, human well-being will be treated as having multiple dimensions. It refers to the degree to which an individual, family, or larger social grouping (e.g. firm, community) can be characterized as being healthy (sound and functional), happy, and prosperous. (Pollnac et al., 2006). The focus here, however, will be on individual well-being, although the determinants of an individual's well-being can include characteristics that include characteristics of family, community, nation, and so forth.

Similar to work done by natural scientists to describe ecological components that represent the system's overall biophysical health, social scientists have created broad categories or domains to draw general distinctions among different components of HWB. Within each domain is a set of subcategories or attributes that identify the specific components of HWB for that domain. There is no one generally agreed upon set of domains and attributes to describe HWB. In reviewing over 22 studies, Hagerty et al (2001) found the following seven domains to be broad enough to encompass most research frameworks: relationships with family and friends; emotional well-being; material-well-being; health; work and productive activity; feeling part of one's community; and personal safety (see also Cummins, McCabe, Romeo, and Gullone, 1994; Cummins, 1996).

The list of potential attributes is even longer, and no comprehensive list exists. Examples of attributes include items such as education; employment; energy; human rights; shelter, housing; health and health care access; income, income distribution, purchasing power; mobility; transportation; infrastructure; governing institutions; social participation; population; reproduction; leisure activities, sports participation and vacation time; spirituality; public safety and crime; traditional activities and cultural responsibilities; and more (Diener and Suh, 1997; Boelhouwer, 1999; Marks, 2007; Costanza, et a l., 2007; Flynn, 2002).

Domains and attributes are concepts that allow researchers to understand and broadly categorize information. Indicators are the actual measures that communicate information about the state of and trends in HWB for a given system. They are most useful when the cost of gathering information about the entire system is high, so that information must be simplified into a set of easily quantifiable attributes that represent the entire system. Indicators have been the subject of considerable discussion in both the natural and social sciences, in disciplines such as economics, sociology, anthropology, psychology, ecology, forestry and many others. Due to the broad array of disciplinary approaches, definitions, and applications, the formulation of indicators varies widely depending on which ‘world view' is applied (Bowen and Riley, 2003). For example, the management community has focused on institutional measures of program performance while the ecological science community has worked to build indicators of the scope and scale of change in natural systems. The social science community has created social indicators to measure trends and changes in social systems.

Social indicators are societal measures that reflect people's circumstances in a given cultural or geographic unit. Land (1983) identifies three primarily uses for social indicators: monitoring (i.e., reporting for policy assessment), tracking (i.e., reporting for public enlightenment), and forecasting. Social indicators can focus on populations of interest such as the elderly, disabled, minorities, or women; or they can be used to track changes in geographic regions There are two types of social indicators for measuring human well-being: objective and subjective indicators (Diener and Suh, 1997; Costanza et al., 2007; Cummins, 2000). Objective indicators are those that can, in principle, be measured and verified in the "public domain," as expressed by Cummins (2000). Examples of objective social indicators include infant mortality, doctors per capita, and longevity (assessed for the health domain); and homicide rates, police per capita, and rates of rape (assessed for the personal safety domain). Objective indicator data can be gathered by observation or other forms of impersonal measurement, or by surveys that seek objective information from individual responses. The key feature of an objective indicator is the perspective: In principle, they measure attributes of human well-being that are publicly visible and have a uniform interpretation across individuals.

Objective social indicators help us understand how specific communities utilize resources or interact with the environment, but they do not measure how people feel about their place or their subjective experience influenced by the health of the environment. Subjective social indicators attempt to measure psychological satisfaction, happiness, and life fulfillment, which are private attributes of HWB in the sense of not being capable of independent observation and verification. By necessity, subjective social indicators are gathered through survey research instruments that ascertain the subjective reality in which people live. Sharpe (1999) describes this approach as "based on the belief that direct monitoring of key social-psychological states is necessary for an understanding of social change and the quality of life." Different domains lend themselves to being measured and tracked by different types of indicators. Material well-being and other basic economic attributes of HWB are amenable to being measured with objective indicators. These are often derived from data gathered by the U.S. Census Bureau or other government agencies. Even these domains, however, have important subjective elements, and so tracking both objective and subjective indicators will provide a more complete understanding of HWB and environmental considerations.

It is important to understand whether a social indicator has an unambiguous relation to HWB at either the individual or aggregate level, or whether it merely describes an attribute of HWB but without such a clear relation. If the first case holds, Land (1983) suggests that the indicator can then be used as a normative indicator, or one that can be directly tied to a social policy goal (Sharpe, 1999). The US Department of Health has defined normative welfare indicators in the following way:

"…a statistic of direct normative interest which facilitates concise, comprehensive and balanced judgments about the condition of major aspects of a society. It is, in all cases, a direct measure of welfare and is subject to the interpretation that if it changes in the 'right' direction, while other things remain equal, things have gotten better, or people are better off. Thus, statistics on the numbers of doctors or policemen could not be social indicators, whereas figures on health or crime rates could be (Land, 1983)."

The use of normative social indicators in this sense requires that society agree about what needs to be improved, that agreement exists on what "improved" means, and that it is meaningful to aggregate the indicators to the level of aggregation at which policy can be defined (Land 1983). Normative social indicators are most useful when indicators are used for policy monitoring, and they can be either objective or subjective in nature.

If an indicator does not have a clear policy relation, it can still be used as a descriptive indicator (Land, 1983), and can again be either objective or subjective in nature. As Land (1983) notes, descriptive social indicators focus on "social measurement and analysis designed to improve our understanding of what the main features of society are, how they interrelate, and how these features and their relationships change." This type of indicator may be related to social policy objectives, but is not restricted to this use (Sharpe 1999). Descriptive social indicators come in many forms, and can vary greatly in the level of abstraction and aggregation, from a diverse set of statistical social indicators to an aggregated index of the state of society.

As should be clear from the discussion above, human well-being is a complex concept, impossible to observe and measure directly, from the viewpoint of an objective observer. Nevertheless, there is broad agreement on important areas such as HWB domains, some of which can be connected to Partnership goals and objectives. Thus, identifying social indicators for the Partnership's efforts is a tractable task, although the basis for selecting a particular set of indicators is still daunting.

Key Points: Human well-being is difficult to define and measure from an objective point of view, but can be categorized in terms of its domains, such as material and emotional well-being, work and productive activity, and personal safety. Indicators connected to these domains can be objective or subjective in nature, and they can be normative (that is having an unambiguous relation to HWB) or descriptive.

The Determinants of Human Well-being

In this section, we consider how research on HWB and its determinants can illuminate the problem of selecting HWB indicators for ecosystem-based management. The focus is on methods that can and have been used to identify economic, social, and sometimes environmental factors that are correlated with and therefore likely to determine (in part) human well-being. These methods provide a way of assessing the connections between ecological and human systems, using human well-being as the metric by which to judge the strength of those links. The methods described below do not span the full set of potential ways of making such an assessment. In later versions of this document, the intent is to add, where warranted, other approaches.

The approach taken here is admittedly a reductionist view of human well-being and its determinants. First, we collapse the multiple domains or dimensions of human well-being into a single measure. While this measure is not observable directly, we use a framework that is based on either subjective, self-reported evaluations or inferred from observable behavior. Second, we assume that HWB can be expressed as a function of measurable, objective circumstances. There may be many other determinants, of course, that are not easily measured or even observable, but the challenge of selecting indicators for HWB can only be met if this second assumption holds.

With these assumptions, we can then formally represent HWB in the following way (Welsch and Kühling, 2009):

$HWB = F(M,X,D,Q,U)$

where HWB is an individual's stated well-being (the measurement of which is discussed below); M is the individual's income; X is a set of community or higher level "macro" factors that help determine HWB; D is a set of individual-level factors that help determine HWB; Q is a set of environmental conditions that determine the individual's HWB.; and U is a set of unobserved (or unmeasured) HWB determinants.

This equation provides a basis for formally and quantitatively assessing the links between a particular environmental quality attribute, Qi, and HWB:

$$dHWB = \frac{\partial F}{\partial Q_i} dQ_i$$

which provides a theoretical construct for evaluating what environmental quality attributes are connected to HWB (i.e., is $\partial F/\partial Q_i > 0$?) and to assess the strength of the connections (i.e., what is the magnitude of $\partial F/\partial Q_i$?) (Welsch and Kühling, 2009).

Below, we consider three general strategies for bringing this equation to life. The first, generally known as life satisfaction or "happiness" studies, starts with direct measurement of HWB and then analyzes objective factors that correlate with that measurement. The other two are different approaches used in economics based on the willingness of individuals to sacrifice one good (usually taken as income) for others, or a "willingness-to-pay" (WTP) approach. The first of these is based on the actual behavior of individuals, either observed directly or inferred through market prices. The second of the WTP approaches is based on the stated preferences of individuals regarding their willingness-to-pay for one situation relative to another. Each of these three approaches uses the equations above in one way or another to derive quantitative estimates of connections between HWB and its determinants.

1. Direct, Subjective Measurement of Human Well-being

The question of an individual's well-being can be addressed by taking a straightforward approach: Ask a person directly. The literature that has built up around this approach is generally known as life satisfaction or "happiness" studies. The types of measures used to assess HWB in this way fall into two categories: (1) measures that reflect an individual's self-reported well-being in a global or holistic sense; and (2) measures that reflect an individual's self reported well-being in the moment (Frey and Stutzer, 2002; Vitarelli, 2010).

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* **Because the different components of *Q* are likely to have different units, it is more likely that this expression would be measured as an *elasticity*, or**

$$e_i = \frac{Q_i}{HWB_i}\frac{\partial HWB}{\partial Q_i}.$$

An elasticity is a unitless number that measures the proportional change in the function with respect to a proportional change in one of its arguments.

This approach and methods to analyze life satisfaction and happiness originated in psychology but have been of found increasing interest to economists. The existence of several long-running, multi-national surveys provide a rich set of data for analysis (Frey and Stutzer, 2002):

- The General Social Surveys, which asks: "Taken all together, how would you say things are these days—would you say that you are very happy, pretty happy, or not too happy?" (Davis, Smith, and Marsden, 2001).
- The World Values Survey, which uses a ten-point scale and asks respondents: "All things considered, how satisfied are you with your life as a whole these days?" (Inglehart et al. 2000).
- The Eurobarometer Surveys, which covers all members of the European Union and asks respondents: "On the whole, are you very satisfied, fairly satisfied, not very satisfied, or not at all satisfied with the life you lead?" (Noll, 2008)

Other approaches use the answers to multiple questions to address life satisfaction, such as the Satisfaction With Life Scale (Diener et al., 1985), which is composed of five questions and rates life satisfaction on a scale from one to seven.

As is the case for all data gathered through surveys, this approach is prone to a host of possible errors. A person's self-reported global well-being can be influenced by moment-to-moment factors such as mood and immediate circumstances; it can also be affected by survey artifacts such as the order and wording of questions, the response scales used, and the selection of information given as context (Frey and Stutzer, 2002). Whether these factors produce systematic biases depends on how the data are used, as the potential problems are muted if their main use is not to compare levels in an absolute sense but rather to seek to identify the determinants of happiness.

With data on self-reported individual well-being, the framework above can be used to discern the determinants of HWB. The true level of HWB is modeled as a latent variable that is related to objective individual, economic, social, and environmental conditions, and the function above (usually in a linear form) can be estimated using ordered probit or logit regression (Welsch and Kühling, 2009). Among the most studied determinants is income (Hsieh 2003, Solberg et al 2002, Vera-Toscano et al 2006, Warr 1999, and many others). Across individuals within a given location, the general (and very robust) result is the people with higher incomes report higher levels of well-being (life satisfaction or happiness) - "income does buy happiness" (Frey and Stutzer, 2002).

Easterlin (1974, 1995, 2001), however, has found that while this result holds cross-sectionally, as incomes rise over time within a given area (such as a nation), everyone's self-reported well-being does not necessarily increase. This result has been supported by laboratory experiments that look at the effects of individuals' relative income on happiness (Smith et al. 1989, Tversky and Griffin 1991). Another interesting result comes from Alesina, et al. (2001), which found a strong negative relation between income inequality and happiness in Europe, but not in the United States. Another area related to income is unemployment, which many studies have shown to have strong, negative effects on well-being (Clark et al. 2001, Di Tella et al. 2001, Graetz 1993, Korpi 1997, Winkelmann and Winkelmann 1998).

Other individual circumstances play a strong role in determining self-reported well-being. A few areas are criminal victimization (Michalos and Zumbo 2000), housing and home-ownership (Diaz-Serrano 2006), and education (Hayo and Seifert 2003). Di Tella et al. (2001) show how inflation and unemployment both affect an individual's well-being; Frey et al. (2009) show how terrorism in France and the British Isles exerts a strong negative effect on subjective well-being; and Frey and Stutzer (2000), in a study of Swiss cantons, show how the institutional right of individual political participation via popular referenda exerts a strong effect on happiness.

This approach has also been used to examine the relations between environmental conditions and subjective well-being, as shown in Table 1 (Welsch and Kühling, 2009; Ferreira and Moro, 2010). While research on measuring subjective HWB directly and exploring its determinants is growing, the literature has not yet expanded to cover the broad set of ecological goals associated with the Partnership's efforts. Nevertheless, these studies and this method provide an interesting perspective on how links between ecological conditions and HWB can be assessed. If changes in these conditions have progressed to the point of having serious impacts on human systems, viewing the impacts through the lens of direct, subjective measurement of HWB would seem a fruitful avenue. Short of such changes, other methods (such as the ones discussed below) would seem more likely to provide a finer grained assessment of the links.

Table 1. Examples of studies using the direct, subjective measurement approach for estimating links between HWB and environmental quality	
Climate	Becchetti et al. (2007) Frijters and van Praag (1998) Rehdanz and Maddison (2005)
Droughts	Carroll et al. (2009)
Air pollution	Welsch (2002) Welsch (2006) Di Tella and MacCulloch (2007) Luechinger (2009)
Airport noise nuisance	van Praag and Baarsma (2005)
Flood hazards	Luechinger and Raschky (2009
Water pollution	Israel and Levinson (2003)

Revealed Preferences Methods

Standard economic theory is based on the assumption that observable choices made by individuals reveal their expected preferences. Individual utility is inferred from behavior, and is in turn used to explain the choices made (see Slesnick 1998 for an extended discussion). Behavior is therefore a way of inferring well-being, in that individuals are assumed to choose actions that are, from an ex ante perspective, the “best,” or the actions that maximize their well-being. Criticisms of this approach, and particularly the equating of utility and well-being, are legion. One of the leading lines is Kahneman (1999; see also Kahneman and Krueger, 2006; Kahneman and Sugden, 2005; Kahneman, Wakker, and Sarin, 1997), who distinguishes between decision utility (which is what economists analyze) and experience utility, which is akin to the moment-to-moment well-being discussed above. He argues that if the two utilities differ in their implications for public policy, experience utility should be favored over decision utility. A common example given to support this stance is one that features smokers: they may decide to have a cigarette (decision utility), yet be better off if they don’t (experience utility) (Read, 2004).

Nevertheless, although the revealed preference approach is not without its problems, it still offers a rich literature from which to draw, at least for the purpose of investigating links between environmental quality and human well-being. Below, we consider three methods that use actual behavior to assess the determinants of HWB: market-based approaches, hedonic analyses, and non-market behavior-based approaches.

Market-based Approaches

The most obvious way of discerning a link between environmental quality and human well-being is to look for environmental "goods and services" in the marketplace. Environmental resources are often inputs to market-based production processes. If so, their value can be measured directly, if the environmental resources are sold in a market; or inferred, if they are not themselves traded but the products they support are. Techniques for estimating the values in these cases are presented in standard benefit-cost textbooks (e.g., Zerbe and Bellas, 2006; Zerbe and Dively, 1994).

For example, Peters et al. (1989) examines the potential market value of non-timber forest products, such as fruits, latex, and tropical medicines, in a hectare of forestland. This value can be measured by calculating the net revenues per hectare from collecting these goods. Other studies use the costs of undesirable environmental change as a way of estimating the potential value of avoiding such change. Yohe et al. (1998) use the market value of land plus the cost of constructing protective sea walls to estimate the potential damage from sea level rise. The economic costs of climate change, and therefore the economic benefits of avoiding climate change, can also be estimated using this market perspective. Climate change will impact energy markets by shifting demand for energy resources, and the value of this shift can be used to infer these costs (Mansur et al., 2008). Similarly, a change in available water for an area through changes in climate can be valued using a demand model of water consumption in a watershed (Hurd et al., 1999).

The existence of markets for ecological goods and services provides an immediate pathway that connects ecological conditions to HWB. For Puget Sound, a potential source of relevant market-based data covers the commercial harvests of finfish and shellfish (Table 2) (Pacific States Marine Fisheries Commission , 2009). The volume of landings and the amount of revenues demonstrates the obvious value of these environmental goods. Exactly how these measures have or would respond to changes in the quality of their supporting habitat and other environmental conditions has not been the subject of systematic study, however.

Table 2. Examples of market landings and revenues in 2008 for species harvested in Puget Sound (Pacific States Marine Fisheries Commission, 2009)

Species common name	Aquaculture		Commercial (Non-tribal)		Commercial (tribal)	
	Landed weight (lbs)	Landed revenue ($)	Landed weight (lbs)	Landed revenue ($)	Landed weight (lbs)	Landed revenue ($)
Geoduck	4,122,429	$25,353,623	2,290,914	$6,188,422	3,197,846	$11,759,146
Chum Salmon			4,196,843	$3,552,228	4,689,451	$3,717,760
Manila Clam	7,149,458	$18,385,757	5,690	$10,811	788,595	$1,268,706
Dungeness Crab			2,837,020	$6,785,143	4,013,664	$10,198,513
Blue or Bay Mussel	2,963,216	$5,293,124			400	$600
Pacific Oyster	2,222,221	$7,498,498	21,238	$84,094	388,746	$1,253,802
Coho Salmon			205,236	$289,293	1,966,139	$3,389,613
Chinook Salmon			180,821	$566,347	1,387,001	$3,613,382

Hedonic Analyses

Market goods often have multiple characteristics but are sold as a bundle. Analyzing such goods to discern the implicit price of each individual characteristic is an approach known as hedonic analysis. An existing house, for example, contains many characteristics that come as a bundle: numbers of bedrooms and bathrooms, square footage, size of lot, type of energy used, and so forth. If the good is fixed to a certain location, the characteristics of the location also become part of the bundle. Again, for an existing house, such location-specific characteristics include the quality of public schools, proximity to jobs, transportation networks, and even environmental amenities, such as air and water quality or proximity to open space. Each of these characteristics is not explicitly priced, yet the price of the house varies systematically with variation in their levels. Two types of bundled goods are analyzed with this approach: housing (or more generally, property) and jobs (wages).

Hedonic property models collect data on the prices of home sales and housing characteristics, which can include environmental quality and amenities. The expectation is that “good” features of a location (e.g., air and water quality) will be reflected by positive implicit prices for those features, while “bad” features (e.g., toxic waste sites) will have negative implicit prices. Hedonic wage models are based on the assumption that a job is a bundle of characteristics, which cover workplace characteristics as well as location-specific characteristics, including environmental quality and amenities. Here, the expected direction of implicit prices is the opposite of that for hedonic property prices. “Good” features will have a negative implicit effect because workers are willing to accept lower wages in locations with such features; “bad” features are associated with higher wages for the opposite reason. Although hedonic wage models are primarily used in environmental economics to value mortality risk, there are some studies that incorporate a broader set of environmental quality measures.

Exactly how one bounds a “location” for hedonic analysis is important. Most studies are limited to urban areas that have well-defined boundaries, or to other geographic units (counties, census blocks, and so forth) that have similarly well-defined boundaries. The characteristics of the bundled good are then taken from the features found within these boundaries. In contrast, Schmidt and Courant (2006) consider proximity to "nice" places (national parks, lakeshores, seashores, and national recreation areas) in an hedonic wage model. They found that amenities outside the metropolitan area generate compensating wage differentials, as workers are willing to accept lower wages to live in proximity to accessible “nice” places.

The hedonic approach has been used to estimate the values, as reflected in property prices or wage levels, for several types of environmental quality attributes, as shown in Table 3. Examples of studies that examine attributes that are more connected to ecological systems are briefly reviewed below:

- Cho et al. (2009) examined amenity values of forest landscapes in the Southern Appalachian Highlands using a hedonic housing-price framework. Their results show that housing prices respond to the size and the density of forest-patches.
- Bin and Polasky (2005) used a hedonic property price method to estimate how wetlands affect residential property values in a rural area. They found that i) a higher wetland percentage within a quarter mile of a property, ii) closer proximity to the nearest wetland, and iii) larger size of the nearest wetland are associated with lower residential property values.
- Poor et al. (2007) investigated the value of ambient water quality throughout a local watershed in Maryland using a hedonic property value model, focusing on total suspended solids and dissolved inorganic nitrogen. Their results indicate that there is a substantial penalty imposed on property prices by higher levels of total suspended solids and dissolved inorganic nitrogen.
- Bark et al. (2009) examined homebuyers' preferences for nearby riparian habitat in the metropolitan Tucson study area and the data incorporated into a hedonic analysis of single family residential house prices. The results indicate that high quality riparian habitat adds value to nearby homes and that instead of indiscriminately valuing "green" open space, nearby homebuyers distinguish between biologically significant riparian vegetation characteristics.
- Bin et al. (2009) used data from the Neuse River Basin in North Carolina to provide empirical evidence on the effect of a mandatory buffer rule on the value of riparian properties. They found that a riparian property generally commanded a premium, but there was no evidence that the mandatory buffer rule had a significant impact on riparian property values when compared with the control group.
- Netusil (2005) uses the hedonic method to examine how environmental zoning and amenities are related to the price of single-family residential properties sold between 1999 and 2001 in Portland, Oregon. The type of environmental zoning and the property's location affected the price effect of environmental zoning, while the type of amenity and its proximity affected a property's sale price.
- Horsch and David (2009) use hedonic analysis to estimate the effects of a common aquatic invasive species--Eurasian water milfoil (milfoil)—on property values across an extensive system of over 170 lakes in the northern forest region of Wisconsin. Their

results indicated that property on lakes invaded with milfoil experienced an average 13% decrease in value after invasion.

- Halstead et al. (2003) applies the hedonic method to estimate the effects of variable milfoil on shoreline property values at selected New Hampshire lakes. Results indicate that property values on lakes experiencing milfoil infestation may be considerably lower than similar properties on uninfested lakes, but that the results are highly sensitive to the specification of the hedonic equation.
- Michael et al. (2000) used the hedonic approach to estimate the value for nine measures of water clarity for lakefront properties in Maine. They found that the value of water clarity varied across these measures, with the differences in implicit prices large enough to potentially affect policy decisions.

Table 3. Examples of studies using the hedonic approach for estimating links between HWB and environmental quality	
Air pollution	Anderson and Crocker (1971) Chattopadhyay (1999) Freeman III (1974) Graves et al. (1988) Harrison Jr. and Rubinfeld (1978) Murdoch and Thayer (1988) Nelson (1978) Nourse (1967), Zabel and Kiel (2000)
Water quality	Boyle et al., (1999) Leggett and Bockstael (2000) Poor et al. (2006) Epp and Al-Ani (1979) Gibbs et al. (2002) Halstead et al. (2003)
Noise	Hall et al. (1978) Nelson, J. P. (1982) O'Byrne et al. (1985) Taylor et al. (1982)
Solid waste sites	Havlicek et al. (1971) Reichert et al. (1992) Thayer and Rahmatian (1992)
Shore erosion protection	Kriesel et al. (1993)
Toxic waste sites	Kiel, K.A. (1995) Kohlhase (1991) Reichert (1997) Smith and Desvousges (1986) Smolen et al. (1992)

Non-market Behavior-based Approaches

For many recreational and other environmental experiences, there is no formal market that can be used to assess their value, either directly or indirectly as is done with the hedonic approach. If the experience requires some form of travel or other behavior that entails a cost (usually in terms of time), however, it is possible to infer how an individual values that experience in terms of their willingness-to-pay. The most common form of this approach is the travel cost method, which uses travel costs and visitation rates to a recreation site to estimate a demand function for that type of recreation (Clawson, 1959; Knetsch, 1963). Similar to the assumptions for hedonic models, the recreation "good" can be a bundle of characteristics, some of which are the environmental features important to the recreational experience. If data are available for visits to multiple sites with varying levels of those features, one can then estimate the contribution of a particular feature to the demand for that recreation, and from this estimate its value (Morey, 1981).

The travel cost method has been widely used to estimate the value of recreation. Loomis (2005) summarizes many of these studies for the purpose of assessing recreation values that could be applied to the U.S. National Forest system. Table 4 presents estimates from Loomis (2005) of seven different types of recreation, drawn from studies conducted in Oregon or Washington. As will be illustrated in the next section, the travel cost and other non-market behavior-based methods have been largely overtaken by the state preference approach. Nevertheless, there are some studies worth noting:

- Murray et al. (2001) estimated the value of reducing beach advisories in Great Lakes beaches located along Lake Erie's shoreline in Ohio. They found that the across all visitors, the average seasonal WTP to encounter one less advisory was approximately $28 per visitor.
- Egan et al. (2009) used a set of water quality measures developed by biologists in a study of recreation visits to 129 lakes in Iowa, and derived estimates of the willingness-to-pay for improvements in the water quality measures. The results demonstrated a significant WTP for water clarity as measured by the Secchi transparency, and that recreational trips decreased as concentrations of nutrients increased.
- Massey et al. (2006) and estimated the benefits of reducing water pollution for recreational fishing when fishing takes place at multiple locations. They found only small impacts from improving water quality conditions in Maryland's coastal bays alone, but that improvements throughout the range of the species could increase abundance and associated beneficial increased catch rates.
- Montgomery and Needelman (1997) also estimated the benefits of reducing water pollution for recreational fishing when fishing takes place at multiple locations. They estimated an annual benefit of $63 per capital per seasons from eliminating toxic contamination from New York lakes and ponds.
- Johnstone and Markandya (2006) derived economic values for river quality indicators, including chemical, biological and habitat-level attributes, by developing a model of angler behavior that linked these attributes to visitation rates. The models could then be used to estimate the welfare associated with marginal changes in river quality.

Table 4. Average recreation values based on studies from Oregon and Washington that use the revealed preference approach (Loomis, 2005)

Activity	Value per day ($2004)	Number of studies
Fishing	$41.98	5
Hiking	$23.98	5
Hunting	$35.27	5
Motorboating	$12.48	1
Swimming	$6.06	1
Wildlife viewing	$35.00	3

Stated Preference Methods

Stated preference methods rely on survey questions that ask individuals to make a choice, describe a behavior, or state directly what they would be willing to pay for specified changes in non-market goods or services. This approach is controversial because in most cases it is not possible to verify independently the answers given to the survey questions, although experimental work has been conducted to investigate this issue (Murphy et al., 2005). Stated preference methods are increasingly used in economic studies of environmental quality because they offer the opportunity to estimate the valuation for anything that can be presented as a credible and consequential choice. Because they do tie willingness-to-pay to a hypothetical act of payment, they do not require observations of actual behavior and so they are the only economic methods that can measure non-use values.

The stated preference method can take the form of a contingent valuation survey, which asks respondents directly about the monetary value of a particular commodity or environmental change (Mitchell and Carson, 1989). A second approach, and one that is increasingly common, is the choice experiment or conjoint analysis approach (Holmes and Adamowicz 2003). This survey method gives respondents a set of hypothetical scenarios, each depicting a bundle of environmental attributes supplied at a given level, where the levels vary across scenarios. Also included (in nearly all cases) is a monetary cost, often characterized as a payment to a fund, a tax, or some other payment mechanism. Respondents are asked to express their preferences by choosing the most preferred alternative, ranking them in order, or rating them on some scale. By examining the tradeoff between the environmental attributes levels and the payment amounts, the willingness-to-pay for the different attributes can be estimated.

Although this approach has focused mainly on environmental economic issues, it has also been used to address other, non-environmental issues, including violent crime (Atkinson et al., 2005);

urban amenities (Howie et al., 2010); broadband service (Tseng and Chiu, 2005); and public transit stop information (Caulfield and O'Mahony, 2009). Cook and Ludwig (2002) examined people's views of policies designed to reduce gun violence using a stated preference model. They asked respondents how they would vote on a policy that was described as having the potential to reduce gun violence by 30 percent. Stated preference questions were used to measure respondents' likelihood of using the high occupancy traffic lanes as a function of the toll level and time savings (Georgia State Road and Tollway Authority, 2005).

Stated preference studies are by far the richest literature for connecting environmental conditions to HWB, at least as measured in terms of individuals' willingness-to-pay. Examples are cited in Table 5, which lists stated preferences studies that have estimated the willingness-to-pay for protecting a species (Richardson and Loomis, 2009). Below, a few of the many other studies are summarized:

- Carson and Mitchell (1993) perform a single comprehensive CV analysis, asking a national random sample of U.S. households to value the change in water quality that results from moving from no pollution control to "swimmable" water quality nationwide. Their best estimate of annual benefits is $(1990) 29.2 billion.
- Lyon and Farrow (1995) assessed the incremental net benefits of additional water pollution control investments beyond 1990. They concluded that these programs could have net benefits less than zero, but significant uncertainties remained.
- Milon, J.W., and D. Scrogin (2005) estimated the benefits of restoring the Greater Everglades ecosystem in Florida. They cast the restoration in terms of ecological functions (water levels) and structural changes (species populations) and found higher WTP for the latter than the former.
- Bell et al. (2003) used a stated preference survey to determine the WTP for a local coho salmon enhancement program in four Washington and Oregon coastal estuaries. They estimate this WTP to range between $37 and $120, depending on a household's income and the type of program.
- Hall et al. (2002) measured the benefit of an improvement in the quality of rocky intertidal zones in southern California resulting from additional regulation enforcement and access limitations. They presented respondents with a hypothetical reduction in illegal collecting and onsite habitat disturbance, which would increase the abundance of intertidal organisms, and found an average WTP of $6 per family-visit.
- Viscusi et al. (2007) used the stated preference approach to estimate values for water quality ratings based on the US Environmental Protection Agency National Water Quality Inventory ratings. They found an average value of $32 for each percent increase in lakes and rivers in the region for which water quality was rated as "Good."
- Banzhaf et al. (2006) quantified the total economic value of ecological improvements to New York's Adirondack Park from a reduction in acid rain. They estimated the WTP for these improvements to range from $48 to $107 annually.

Table 5. Examples of studies that use the stated preference approach to estimate the economic non-use value of a species (Richardson and Loomis, 2008)

Arctic grayling	Duffield and Patterson (1992)
Atlantic salmon	Stevens et al. (1991)
Bald eagle	Boyle and Bishop (1987) Stevens et al. (1991) Swanson (1996)
Bighorn sheep	King et al. (1988)
Blue whale	Hageman (1985)
Bottlenose dolphin	Hageman (1985)
Gray whale	Hageman (1985) Loomis and Larson (1994)
Gray wolf	Duffield (1991, 1992) Duffield et al. (1993) Chambers and Whitehead (2003)
Humpback whale	Samples and Hollyer (1989)
Mexican spotted owl	Loomis and Ekstrand (1997) Giraud et al. (1999)
Monk seal	Samples and Hollyer (1986)
Northern spotted owl	Rubin et al. (1991) Hagen et al. (1992)
Northern elephant seal	Hageman (1985)
Peregrine falcon	Kotchen and Reiling (2000)
Red-cockaded woodpecker	Reaves et al. (1994)
Riverside fairy shrimp	Stanley (2005)
Salmon	Olsen et al. (1991) Loomis (1996) Layton et al. (2001) Bell et al. (2003)
Sea otter	Hageman (1985)
Silvery minnow	Berrens et al. (1996)
Squawfish	Cummings et al. (1994)
Steller sea lion	Giraud et al. (2002)
Striped shiner	Boyle and Bishop (1987)
Whooping crane	Bowker and Stoll (1988)
Wild Turkey	Stevens et al. (1991)

Summary

Given the flexibility of the stated preference approach, it is tempting to ignore the first two methods – direct, subjective HWB measurement and revealed preference approaches – and focus on the stated preference approach as the most fruitful, at least in terms of ongoing and future research. That approach can be difficult to apply for ecological systems, however, because presenting information on such systems in the context of a survey can be problematic (Boyd and Krupnick, 2009). For the first two methods, an individual does not need to understand or even be aware of entire system that connects ecological conditions and well-being. These methods are based on the actual experience of these conditions, however, because they use objective measurements of the "real" conditions as the basis for analysis. For stated preference surveys, the connections are explored by giving individuals information about various scenarios, which inevitably decompose the environment into a limited set of abstract conditions. This means that respondents do not experience the full set of "real" conditions, and so are likely to "fill in the gaps" in ways that present problems for gathering useful data (Boyd and Krupnick, 2009).

In any case, there is much more work to be done to relate changes in environmental conditions to changes in human well-being. (Stiglitz et al. 2009). One must be careful in drawing conclusion from the current literature, as the absence of evidence documenting the strength of a connection should never be taken as evidence of the absence of such a connection. Nevertheless, documenting such absences can identify potentially important areas for future research.

Key Points: Although human well-being cannot be observed directly, there are methods to assess the determinants of human well-being. Research has utilized these methods to investigate the strength of connections between economic, social, and environmental factors and HWB. There is still much work to be done, however, in documenting these connections, particularly those covering environmental factors in general and for Puget Sound in particular.

Linking Biophysical and HWB Indicators

In this last section, we briefly present a framework for establishing connections between potential indicators of ecosystem biophysical conditions and human well-being in Puget Sound. The framework also provides a way of characterizing existing and future studies and data that are relevant to an element of the set of potential HWB indicators.

1. Connections between biophysical and human-based indicators

Just as the Partnership's biophysical goals can be linked to human well-being, so too can biophysical indicators. In some cases, the component tracked by a biophysical indicator is directly connected to HWB. A component such as a species, for example, can be valued for its existence, even without any direct consumptive use (e.g., harvest) or non-consumptive activity (e.g., wildlife viewing). Some of the species in Table 4, for example, have little value other than this existence value, and so a measure of some aspect of that species' biological status could serve both as a biophysical indicator and as a normative indicator of human well-being. Estimates of WTP drawn from state preference studies that then measure existence value are one way of gauging the importance of such an ecological component. This provides a means of identifying a potentially useful indicator, independent of its qualities as a biophysical indicator.

At the other extreme, human well-being is sometimes derived purely from the direct consumption or harvest of an ecological component. The level and value of that use can be used as a normative HWB indicator, easily expressed in dollars if the use takes place in a market setting. In such a case, an indicator that tracks the actual level of consumption or harvest provides information on actual HWB, while an indicator that tracks the biological status of the ecological component provides information on potential future HWB.

This case presents an interesting complication that illustrates some of the nuances involved in introducing HWB into ecosystem-based management. Fishing provides an example relevant to Puget Sound. The harvest of a fish population is an activity that supports HWB, and so an indicator based on harvest levels is one that faithfully tracks HWB. If the harvest rate is unsustainably high, however, an indicator that tracks the status of the fish population will trend downward, which seemingly indicates a decline in HWB.

(For the purposes of this simple example, we assume that the fishery is "mature" in that the initial stock is at or below the level that would produce the maximum sustainable yield or growth rate. In that case, a harvest level greater than the growth rate is one that will lower the stock size and its growth rate, accelerating the stock's decline.)

How should these conflicting signals be interpreted? If a conservation action consists of rebuilding the fish population with a period of lowered harvest levels, both indicators will accurately reflect the effects of this action on HWB. In Figure 8 (top panel), the harvest level is initially above the sustainable level for the initial stock size, which we assume is the desired or target population level. HWB is correspondingly high, but not at a level that can be sustained indefinitely. At some point, restrictions on harvest are imposed for the purpose of rebuilding the stock. These restrictions reduce the current level of HWB, which then increases assuming the rebuilding period at some point allows harvest to increase gradually. Finally, harvest is maintained at a sustainable level after the stock is rebuilt, and (in this simplistic world) can be maintained at that level indefinitely.

The current level of HWB faithfully tracks the harvest level throughout these periods, and so a normative HWB indicator can be developed based on annual harvests. At the same time, the fish population dynamics foretell future HWB. In Figure 8 (bottom panel), the stock size decreases during the period of overharvesting to levels significantly below its initial, target level. During the rebuilding period, it increases, eventually reaching the target level, where it can be maintained indefinitely as long as the harvest level is sustainable. Again, these movements are faithful predictors of future HWB, and so a normative HWB indicator can be based on its level, recognizing that the information embedded in such an indicator is partly dependent on how the system is managed. This example underscores the complexities in interpreting biophysical indicators in terms of HWB, given the dynamic nature of ecosystems and the potential of natural capital to support current and future HWB.

In other cases, connections exist between ecological and human systems that support HWB along even more complicated pathways. Understanding these pathways is important to identifying potential indicators, evaluating their qualities, and understanding how to relate changes in their levels to changes in HWB. For example, the harvest example illustrated in Figure 8 focuses only on the HWB derived from the connection between a fish population and its harvest by humans. Such a population can be valued along multiple pathways, however, some of which are complementary to harvest while others potentially involve tradeoffs.

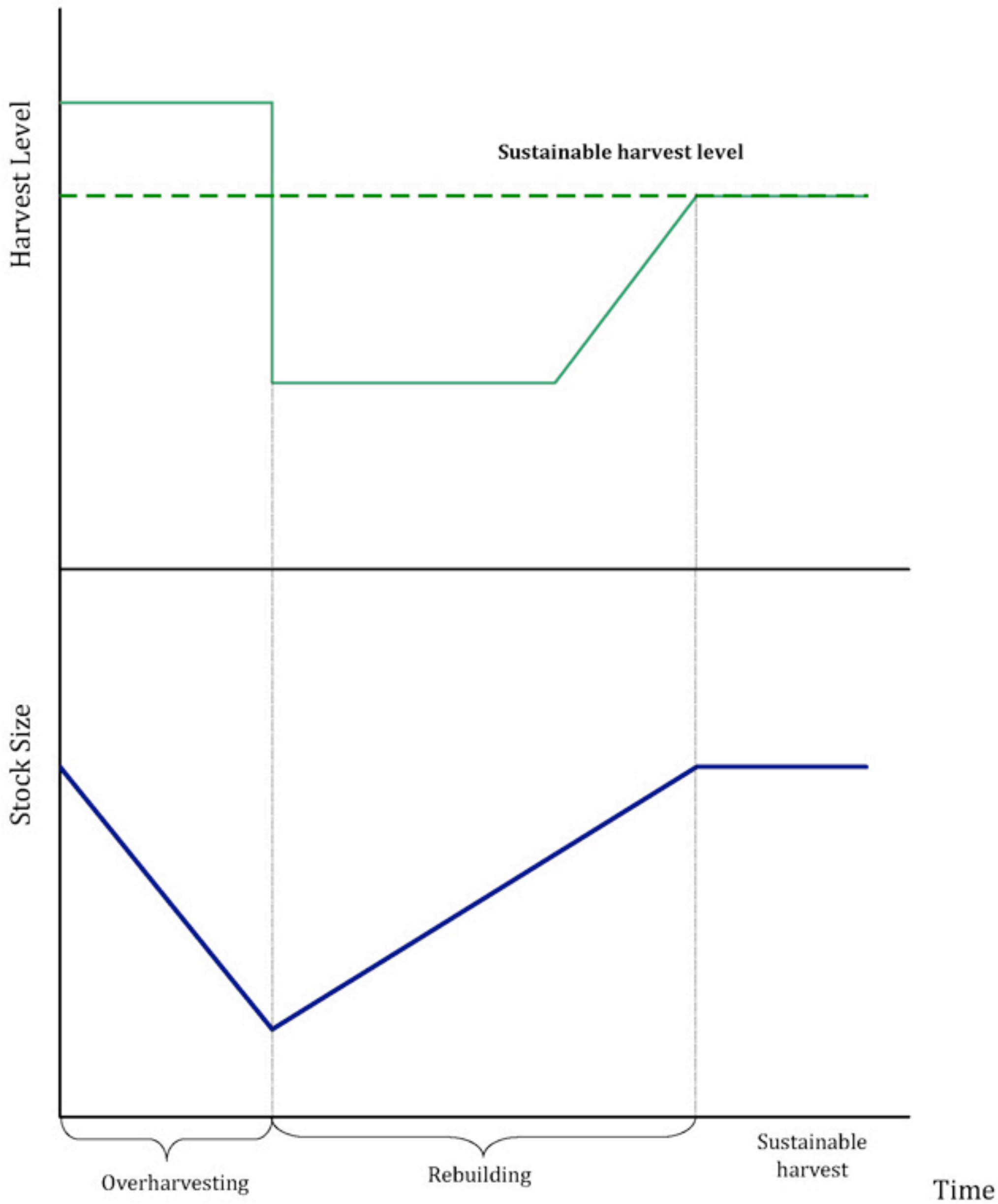

Figure 8 (upper panel): After a period of exceeding the sustainable level, harvest is reduced to allow the fish population to rebuild, and then is gradually increased to its sustainable level. These changes are mirrored by changes in current HWB.
Figure 8 (lower panel): Changes in the stock size accurately track potential future harvests and so can be used as an indicator for future HWB.

For example, Puget Sound coho salmon populations provide opportunities for recreational and commercial fisheries, some of which are conducted by Puget Sound tribes (Pacific Marine Fisheries Council, 2010, Tables B-39 and B-41). They are also prey for bald eagles (Stinson et al., 2007), an iconic species that has considerable economic value for wildlife viewing and existence value (Boyle and Bishop, 1987; Stevens et al., 1991; Swanson, 1996). In the Skagit

River basin, coho populations have experienced a loss in spawning and rearing habitat due to economic activities such as flood control, agriculture, and other activities (Stinson et al., 2007). Focusing on agriculture, we note that the Partnership has identified it as a "Low Threat" to ecosystem health (Puget Sound Partnership, 2009c). The Partnership has also identified "locally-grown food" in its Action Agenda as part of its five primary objectives, under the qualification that its production be "consistent with ecosystem protection" (Puget Sound Partnership, 2008). The cost and quality of agricultural production is an obvious contributor to HWB, as evidenced by its market value; moreover, there is some evidence that locally-produced food can command a higher WTP, other characteristics constant Darby et al., 2008). All of these connections create a complex set of pathways between potential biophysical and human-based indicators, and between those indicators and potential management actions (Figure 9).

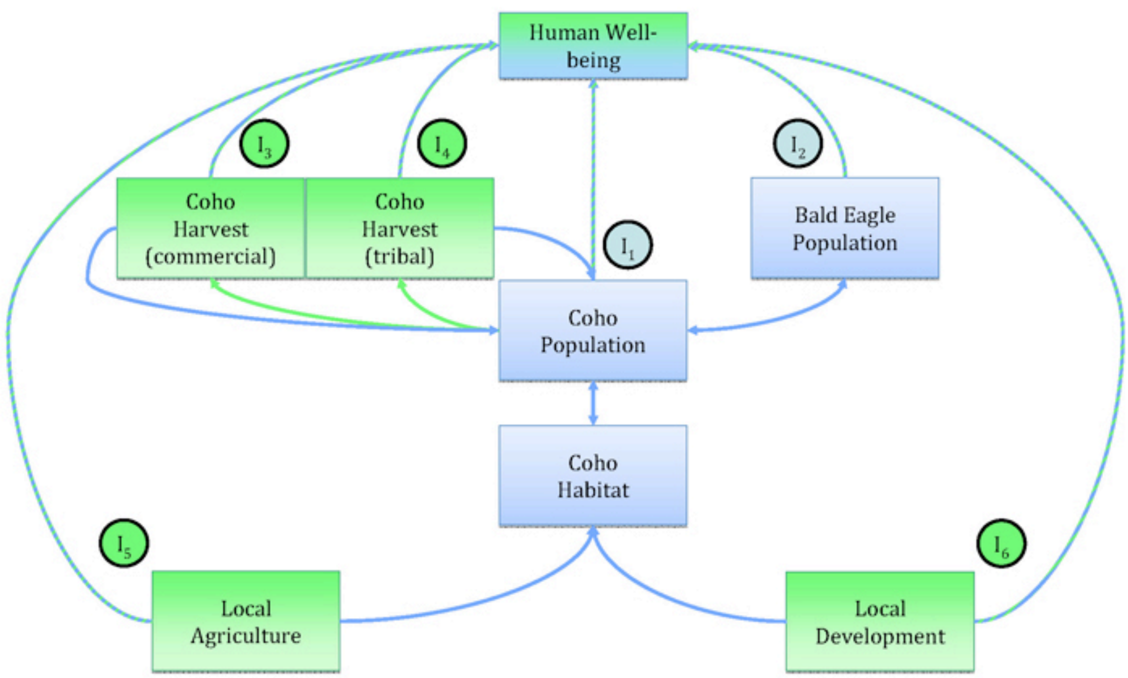

Figure 9: An example of the connections between biophysical and human-based components of the Puget Sound ecosystem, and between those components and human well-being. Identifying these connections can facilitate the identification and evaluation of biophysical and human well-being indicators.

In this system, HWB indicators could be based on

- Coho and bald eagle populations (I1 and I2). Bell et al. (2003) used a stated preference survey to determine the WTP for a local coho salmon enhancement program in four Washington and Oregon coastal estuaries. They estimate this WTP to range between $37 and $120, depending on a household's income and the type of program. Swanson (1996) used a stated preference survey to determine the WTP of visitors to the Skagit River Bald Eagle Natural Area for bald eagle preservation. She found that visitors were willing to pay up to $350 for a 3005 increase in their population.

- Commercial Puget Sound coho harvest (all sources) and commercial, ceremonial, and subsistence tribal Puget Sound coho harvest levels (I3 and I4). As noted before in Table 2, Puget Sound coho populations are a valuable market commodity.
- Locally-based agricultural production (I5). Darby et al. (2008) used a stated preference survey to address whether consumers place a premium on "local" food distinct from other agricultural characteristics such as product freshness. They found that "local" does command a premium but found no difference between "in state" and "nearby" as the relevant geography for "local".
- Local development (I6). Because human well-being is supported by myriad forms of capital, not just natural capital (Millennium Ecosystem Assessment, 2005), measuring the contribution of land development to HWB and utilizing an appropriate indicator are important for EBM. This is an area for future work.

For broader purposes, one could use this approach for identifying connections and potential indicators to refine the Partnership's development of objectives and performance measures based on the Open Standards framework and its results chains (Puget Sound Partnership, 2009d).

Summary

Assessing the strength of connections between HWB and biophysical or human-based components of the ecosystem provides some guidance for EBM, then, in several ways. First, where sufficient evidence exists to indicate the strength of a connection, using any of the approaches described in the previous section, such evidence can highlight potential indicators associated with relatively strong connections. Second, the evidence can at least give some insights into the overall effect on HWB in cases where proposed management actions have multiple effects and potential tradeoffs. The evidence might indicate where such tradeoffs are likely to be "one-sided," in the sense of one value or connection being significantly stronger than any other; or it might indicate where such tradeoffs might be "closer," in that they involve multiple connections with some value but which move in opposite directions in response to a proposed action. And finally, collecting and cataloging evidence of this sort can highlight the (unfortunately many) areas where evidence is sparse, particularly for the connections among biophysical conditions, human behavior and values, and overall human well-being in the Puget Sound region. This can help set priorities for future social science research to support the Puget Sound Partnership's mission.

Key Points: The evidence on connections between environmental conditions and human well-being can be used to identify and evaluate potential indicators for the Puget Sound Partnership. Some biophysical indicators can also serve as human well-being indicators, or can be used in conjunction with HWB indicators to which they are connected. Evidence drawn from studies on HWB and environmental conditions can be used to assess the potential importance of the connections between the two, and so provide the Partnership with guidance on choosing relevant indicators.

References

Alesina, A., R. di Tella, and R. MacCulloch, Inequality and happiness: Are Europeans and Americans different?, NBER working paper 8198, 2001.

Anderson, R. J., Jr. and T. D. Crocker, Air pollution and residential property values, Urban Studies, 1971. 8: 171-180.

Atkinson, G., A. Healey, and S. Mourato, Valuing the costs of violent crime: a stated preference approach, Oxford Economic Papers, 2005. 57: 559-585.

Banzhaf, S., D. Burtraw, D. Evans, and A. Krupnick, Valuation of natural resource improvements in the Adirondacks, Land Economics, 2006. 82: 445–464.

Bark, R. H., D.E. Osgood, B. Colby, G. Katz, and J. Stromberg, Habitat preservation and restoration: Do homebuyers have preferences for quality habitat? Ecological Economics, 2009. 68: 1465-75

Becchetti, L., S. Castriota, and D.A. Londono Bedoya, Climate, happiness and the Kyoto protocol: someone does not like it hot, Departmental Working Paper No. 247, Tor Vergata University, 2007.

Bell, K.P, D. Huppert and R.L. Johnson, Willingness to pay for local coho salmon enhancement in coastal communities, Marine Resource Economics, 2003. 18: 15–31.

Berrens R.P., P. Ganderton and C. Silva, Valuing the protection of minimum instream flows in New Mexico, Journal of Agricultural and Resource Economics, 1996. 21:294–309.

Bin, O., and S. Polasky, Evidence on the amenity value of wetlands in a rural setting, Journal of Agricultural and Applied Economics, 2005. 37: 589-602.

Bin, O., C. Landry, and G.F. Meyer, Riparian buffers and hedonic prices: A quasi-experimental analysis of residential property values in the Neuse River basin, American Journal of Agricultural Economics, 2009. 91: 1067-79.

Boelhouwer, J., and I. Stoop, Measuring well-being in the Netherlands: The SCP index from 1974 to 1997, Social Indicators Research, 1999. 48:51-75.

Bowen, R.E., and C. Riley, Socio-economic indicators and integrated coastal management, Ocean and Coastal Management, 2003. 46: 299-312.

Bowker J.M., and J.R. Stoll, Use of dichotomous choice nonmarket methods to value the whooping crane resource, American Journal of Agricultural Economics, 1988. 70: 372–381.

Boyd, J., and A. Krupnick, The definition and choice of environmental commodities for nonmarket valuation, Resources for the Future Discussion Paper 09-35, 2009.

Boyle, K. and R. Bishop, Valuing wildlife in benefit–cost analysis: a case study involving endangered species, Water Resources Research, 1987. 23: 943–950.

Boyle, K. J., P. J. Poor and L. O. Taylor, Estimating the demand for protecting freshwater lakes from eutrophication, American Journal of Agricultural Economics, 1999. 81: 1118-1122.

C.M. Chambers and J. Whitehead, A contingent valuation estimate of the benefits of wolves in Minnesota, Environmental and Resource Economics, 2003. 26: 249–267.

Carr, E.R., et al., Applying DPSIR to Sustainable Development, International Journal of Sustainable Development and World Ecology, 2007. 14 (6) 543-555.

Carroll, N., Frijters, P. and Shields, M.A., Quantifying the costs of drought: new evidence from life satisfaction data, Journal of Population Economics, 2009, 22: 445-461.

Carson, R. and R. Mitchell, The value of clean water: the public's willingness to pay for boatable, fishable, and swimmable quality water, Water Resources Research, 1993. 29: 2445-2454.

Caulfield, B., and M. O'Mahony, A stated preference analysis of real-time public transit stop information, Journal of Public Transportation, 2009. 12: 1-20.

Chattopadhyay, S., Estimating the demand for air quality: new evidence based on the chicago housing market, Land Economics, 1999. 75, 22-38.

Cho, S., S.G. Kim, R.K. Roberts, and S. Jung, Amenity values of spatial configurations of forest landscapes over space and time in the southern Appalachian Highlands, Ecological Economics, 2009. 68: 2646-57

Clark, A.E., Y. Georgellis, and P. Sanfey, Scarring: The psychological impact of past unemployment, Economica, 2001. 68: 221–41.

Clawson, M. Methods of measuring the demand and value of outdoor recreation, Resources for the Future, Reprint 10, 1959. Washington, DC

Cook; P.J., and J. Ludwig, The Costs of Gun Violence against Children, The Future of Children, 2002. 12: 86-99.

Costanza R, Fisher B, Ali S, Beer C, Bond L, Boumans R, Danigelis N, Dickinson J, Elliott C, Farley, J, Gayer DE, Glenn LM, Hudspeth T, Mahoney D, McCahill L, McIntosh B, Reed B, l, Rizvi, SAT, Rizzo DM, Simpatico T, and Snapp R. 2007. Quality of life: An approach integrating opportunities, human needs, and subjective well-being, Ecological Economics, 1: 267-276.

Cummings R., P.G. Cummings, and T. McGuckin, Substitution effects in CVM values, American Journal of Agricultural Economics, 1994. 76: 205–214.

Cummins, R. A., M. P. McCabe, Y. Romeo, and E. Gullone, The comprehensive quality of life scale: Instrument development and psychometric evaluation on tertiary staff and students, Educational and Psychological Measurement, 1994. 54: 372-382.

Cummins, R., Objective and subjective quality of life: An interactive model, Social Indicators Research, 2000. 52: 55-72

Cummins, R.A., The domains of life satisfaction: An attempt to order chaos, Social Indicators Research, 1996. 38: 303-328.

Darby, K., M.T. Batte, S. Ernst, and B. Roe, Decomposing Local: A Conjoint Analysis of Locally Produced Foods, American Journal of Agricultural Economics, 2008. 90: 476-486.

Dasgupta, P., Human well-being and the natural environment, Oxford: Oxford University Press, 2001.

Davis, J.A., T. Smith, and P.V. Marsden, General social survey, 1972– 2000: Cumulative codebook, Storrs, CT: Roper Center for Public Opinion Research. 2001.

Di Tella, R. and R.J. MacCulloch, Gross national happiness as an answer to the Easterlin paradox?, Journal of Development Economics, 2007. 16: 22–42.

Di Tella, R., R. MacCulloch, and A. Oswald, A., Preferences over inflation and unemployment: evidence from surveys of happiness, American Economic Review, 2001. 91:335–341.

Diaz-Serrano, L., Housing satisfaction, homeownership and housing mobility: A panel data analysis for twelve EU countries, IZA Discussion Papers 2318, 2006. Institute for the Study of Labor (IZA).

Diener, E. and E. Suh, Measuring quality of life: Economic, social, and subjective indicators, Social Indicators Research, 1997. 40:189-216.

Diener, E., J. Horwitz and R. A. Emmons, Happiness of the very wealthy, Social Indicators, 1985. 16: 263–274.

Duffield, J., An economic analysis of wolf recovery in Yellowstone: park visitor attitudes and values. In: J. Varley and W. Brewster, Editors, Wolves for Yellowstone?, National Park Service, Yellowstone National Park, 1992.

Duffield, J., Existence and non-consumptive values for wildlife: application of wolf recovery in Yellowstone National Park, W-133/Western Regional Science Association Joint Session. Measuring Non-Market and Non-Use Values. Monterey, CA, 1991.

Duffield, J. and D. Patterson, Field testing existence values: comparison of hypothetical and cash transaction values. In: B. Rettig, Editor, Benefits and costs in natural resource planning, 5th Report. W-133 Western Regional Research Publication, Compiler, Dept. of Agricultural and Resource Economics, Oregon State University, Corvallis, OR, 1992.

Duffield, J., D. Patterson, and C. Neher, Wolves and people in Yellowstone: a case study in the new resource economics. Report to Liz Claiborne and Art Ortenberg Foundation. Department of Economics, University of Montana, Missoula, MT, 1993.

Easterlin, R., Does economic growth improve the human lot? Some empirical evidence. In P.A. David and M.W. Reder (eds), Nations and Households in Economic Growth: Essays in Honour of Moses Abramovitz New York: Academic Press, 1974, pp. 89–125.

Easterlin, R. A., Will raising the income of all increase the happiness of all?, Journal of Economic Behavior and Organization, 1995. 27: 35–47.

Easterlin, R.A., Income and happiness: towards a unified theory, Economic Journal, 2001. 111: 465–84.

Egan, K., J. Herriges, C. Kling, and J. Downing, Valuing Water Quality as a Function of Water Quality Measures, American Journal of Agricultural Economics, 2009. 91: 106-123.

Endter-Wada, J., D. Blahna, R. Krannich, and M. Brunson. A Framework For Understanding Social Science Contributions To Ecosystem Management. Ecological Applications, 1998. 8: 891–904.

Epp, D.J., and K. S. Al-Ani. The Effect of Water Quality on Rural Nonfarm Residential Property Values, American Journal of Agricultural Economics, 1979. 61: 529-534.

Ferreira, S., and M. Moro. On the Use of Subjective Well-Being Data for Environmental Valuation. Environmental Resources and Economics (online at http://www.springerlink.com/content/p8t67q8320476362/ , accessed 16 March 2010)

Flynn P, D. Berry, and T. Heintz, Sustainability and quality of life indicators: Toward the integration of economic, social and environmental measures. Indicators: The Journal of Social Health, 2002. 1: 19-39.

Freeman III, A. M., On Estimating Air Pollution Control Benefits from Land Value Studies, Journal of Environmental Economics and Management, 1974. 1: 277-288.

Frey B.S., and A. Stutzer, What Can Economists Learn from Happiness Research? Journal of Economic Literature, 2002. 40: 402–435

Frey, B.S., S. Luechinger, and A. Stutzer, The life satisfaction approach to valuing public goods: The case of terrorism, Public Choice, 2009. 138: 317-345.

Frijters, P. and B. van Praag, The effects of climate on welfare and well-being in Russia, Climate Change, 1998. 39: 61–81.

Georgia State Road and Tollway Authority, Final Report, NuStats, Interstate 75 Stated Preference Survey, Austin, Texas, 2005.

Gibbs, J.P., J.M. Halstead, K.J. Boyle, and J. Huang, An hedonic analysis of the effects of lake water clarity on New Hampshire Lakefront properties, Agricultural and Resource Economics Review, 2002. 31: 39–46.

Giraud K., J. Loomis, and R. Johnson., Internal and external scope in willingness-to-pay estimates for threatened and endangered wildlife, Journal of Environmental Management, 1999. 56: 221–229.

Giraud, K., B. Turcin, J. Loomis, and J. Cooper, Economic benefit of the protection program for the stellar sea lion, Marine Policy, 2002. 26: 451–458.

Graetz, B., Health Consequences of Employment and Unemployment: Longitudinal Evidence for Young Men and Women, Social Science and Medicine, 1993. 36: 715–724.

Graves, P., J.C. Murdoch, M.A. Thayer, and D. Waldman, The Robustness of Hedonic Price Estimation: Urban Air Quality, Land Economics, 1998. 64: 220-233.

Hageman, R., Valuing marine mammal populations: benefit valuations in a multi-species ecosystem, Administrative Report LJ-85-22, Southwest Fisheries Center, National Marine Fisheries Service, La Jolla, CA, 1985.

Hagen, D., J. Vincent and P. Welle, Benefits of preserving old-growth forests and the spotted owl, Contemporary Policy Issues, 1992. 10: 13–25.

Hagerty, M.R., R. Cummins, A.L. Ferriss, K. Land, A.C. Michalos, M. Peterson, A. Sharpe, J. Sirgy, and J. Vogel J. Quality of life indexes for national policy: review and agenda for research, Social Indicators Research, 2001. 55: 1-96.

Hall, D.C., J.V. Hall, and S.N. Murray, Contingent Valuation of Marine Protected Areas: Southern California Rocky Intertidal Ecosystems, Natural Resource Modeling, 2002. 15: 335-368.

Hall, F.L., B.E. Breston, and S.M. Taylor, Effects of Highway Noise on Residential Property Values, Transportation Research Record, 1978. 686: 38-43.

Halstead, J.M., J. Michaud, S. Hallas-Burt, and P. Gibbs, Hedonic analysis of effects of a nonnative invader (Myriophyllum heterophyllum) on New Hampshire (USA) lakefront properties, Environmental Management, 2003. 32: 391-398.

Harrison, Jr., D., and D.L. Rubinfeld, Hedonic housing prices and the demand for clean air. Journal of Environmental Economics and Management, 1978. 5: 81–102.

Havlicek, Jr., J., R. Richardson, and L. Davies, Measuring the Impacts of Solid Waste Disposal Site Location on Property Values, American Journal of Agricultural Economics, 1971. 53: 869.

Hayo, B. and S. Wolfgang, Subjective Economic Well-being in Eastern Europe, Journal of Economic Psychology, 2003. 24: 329-348.

Holmes, T.P., and W.L. Adamowicz. Attribute-based methods. In: A Primer on Nonmarket Valuation, edited by P.A. Champ K.J. Boyle and T.C. Brown. Dordrecht: Kluwer Academic Publishers, 2003.

Horsch, E.J., and D.J. Lewis, The Effects of Aquatic Invasive Species on Property Values: Evidence from a Quasi-experiment. Land Economics, 2009. 85: 391-409.

Howie, P. S.M. Murphy, and J. Wicks, An Application of a Stated Preference Method to Value Urban Amenities. Urban Studies, 2010. 47: 235-256.

Hsieh, C.M., Income, Age and Financial Satisfaction. International Journal of Aging & Human Development, 2003. 56: 89-112.

Hurd B., J.M. Callaway, J.B. Smith, and P. Kirshen, Economic effects of climate change on US water resources. In: R. Mendelsohn and J. Neumann, eds., The Impact of Climate Change on the United States Economy. Cambridge, UK: Cambridge Univ. Press , 1999, pp. 133–77.

Inglehart, Ronald et al. World Values Surveys and European Values Surveys, 1981–84, 1990–93, 1995–97, Ann Arbor: Institute for Social Research, 2000.

Israel, D. and A. Levinson, Examining the relationship between household satisfaction and pollution. Mimeo, Indiana State University, 2003.

Johnstone, C., and A. Markandya, Valuing river characteristics using combined site choice and participation travel cost models. Journal of Environmental Management, 2006. 80: 237–247

Justus, J., M. Colyvan, H. Regan, and L. Maguire, Buying into conservation: intrinsic versus instrumental value, Trends in Ecology and Evolution, 2009. 24:187–191.

Kahneman, D., Objective happiness. In D. Kahneman, E. Diener and N. Schwarz (eds), Well Being: The Foundations of Hedonic Psychology. New York: Russell Sage Foundation, 1999.

Kahneman, D., and A.B. Krueger, Developments in the measurement of subjective well-being, Journal of Economic Perspectives, 2006. 20: 3–24.

Kahneman, D. and R. Sugden, Experienced utility as a standard of policy evaluation, Environmental and Resource Economics, 2005. 32: 161–181.

Kahneman, D., P.P. Wakker, and R. Sarin, Back to Bentham? Explorations of experienced utility, Quarterly Journal of Economics, 1997. 112: 375–405.

Kiel, K.A., Measuring the Impact of the Discovery and Cleaning of Identified Hazardous Waste Sites on House Values, Land Economics, 1995. 71: 428-435.

King D., D. Flynn and W. Shaw, Total and existence values of a herd of desert bighorn sheep, Benefits and Costs in Natural Resource Planning, Interim Report, Western Regional Research Publication W-133, University of California, Davis, CA, 1988.

Knetsch J.L., Outdoor recreation demands and benefits, Land Economics, 1963. 39: 387–396.

Kohlhase, J.E., The Impact of Toxic Waste Sites on Housing Values, Journal of Urban Economics, 1991. 30: 1-26.

Korpi, T., Is Well-Being Related to Employment Status? Unemployment, Labor Market Policies and Subjective Well-Being among Swedish Youth, Labour Economics, 1997. 4: 125–47.

Kotchen, M., and S. Reiling, Environmental attitudes, motivations, and contingent valuation of nonuse values: a case study involving endangered species, Ecological Economics, 2000. 32: 93–107.

Kriesel, W., C.E. Landry and A. Keeler, Coastal erosion management from a community economics perspective: The feasibility and efficiency of user fees, Journal of Agricultural and Applied Economics, 2005. 37; 451-461.

Land, K.C., Social indicators, Annual Review of Sociology, 1983. 9: 1-26.

Layton D., G. Brown, and M. Plummer, Valuing Multiple Programs to Improve Fish Populations, Washington State Department of Ecology, Olympia, WA, 2001.

Leggett, C. G., and N. E. Bockstael, Evidence of the Effects of Water Quality on Residential Land Prices Journal of Environmental Economics and Management, 2000. 39: 121-144.

Liu, J., T. Dietz, S. Carpenter, M. Alberti, C. Folke, E. Moran, A. Pell, P. Deadman, T. Kratz, J. Lubchenco, E. Ostrom, Z. Ouyang, W. Provencher, C. Redman, S. Schneider, and W. Taylor, Complexity of Coupled Human and Natural Systems, Science, 2007. 317: 1513-1516.

Loomis, J.B., Measuring the economic benefits of removing dams and restoring the Elwha river: results of a contingent valuation survey. Water Resources Research, 1996. 32: 441–447.

Loomis, J.B., and E. Ekstrand, Economic benefits of critical habitat for the Mexican spotted owl: a scope test using a multiple-bounded contingent valuation survey. Journal of Agricultural and Resource Economics, 1997. 22: 356–366.

Loomis, J.B., and D. Larson, Total economic values of increasing gray whale populations: results from a contingent valuation survey of visitors and households, Marine Resource Economics, 1994. 9: 275–286.

Loomis, J., Updated outdoor recreation use values on national forests and other public lands. Gen. Tech. Rep. PNW-GTR-658. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station., 2005, 26 p.

Luechinger, S. and P. Raschky, Valuing flood disasters using the life satisfaction approach, Journal of Public Economics, 2009. 93: 620-633.

Luechinger, S., Valuing Air Quality Using the Life Satisfaction Approach. Economic Journal, 2009. 119: 482-515.

Lyon, R.M., and S. Farrow, An economic analysis of CleanWater Act issues. Water Resources Research, 1995. 31:213–223.

Mansur, E., R. Mendelsohn, and W. Morrison, A discrete continuous model of energy: measuring climate change impacts on energy, Journal of Environmental Economics and Management, 2008. 55:175–93

Marks, E., M.D. Cargo, and M. Daniel, Constructing a health and social indicator framework for indigenous community health research. Social Indicators Research, 2007. 82:93-110.

Massey, D.M., S.C. Newbold, and B. Gentner, Valuing water quality changes using a bioeconomic model of a coastal recreational fishery. Journal of Environmental Economics and Management , 2006. 52: 482–500.

McGillivray, M., and M. Clarke, Understanding Human Well-being, Bookwell Publications, New Delhi, India, 2008.

Michael, H.J., K.J. Boyle, and R. Bouchard, Does the Measurement of Environmental Quality Affect Implicit Prices Estimated from Hedonic Models? Land Economics, 2000. 76: 283-298.

Michalos, A.C., and B.D. Zumbo, Criminal victimization and Quality-of-Life, Social Indicators Research, 2000. 50: 245-295.

Millenium Ecosystem Assessment, Linking ecosystem services and human well-being, In: Ecosystem and Human Well-being: Multiscale Assessment. Capistrano, D., C.K. Samper, M.J. Lee, and C. Raudsepp-Hearne, eds. Island Press, Washington, D.C., 2005, pp. 45–60.

Millennium Ecosystem Assessment, Ecosystems and Human Well-being: A Framework for Assessment, Island Press, Washington, D.C., 2003.

Milon, J.W., and D. Scrogin, Latent Preferences and Valuation of Wetland Ecosystem Restoration. Ecological Economics, 2005. 56: 162-175.

Mitchell, R.C., and R.T. Carson, Using Surveys to Value Public Goods: The Contingent Valuation Method, Resources for the Future, Washington D.C., 1989.

Montgomery M., and M. Needelman, The welfare effects of toxic contamination in freshwater fish, Land Economics, 1997. 73:211–223.

Morey, E., The demand for site-specific recreation activities: a characteristics approach, Journal of Environmental Economics and Management, 1981. 8: 345–371.

Murdoch, J.C., and M.A. Thayer, Hedonic Price Estimation of Variable Urban Air Quality, Journal of Environmental Economics and Management, 1988. 15: 143-146.

Murphy, J.J., P.G. Allen, T.H. Stevens, and D. Weatherhead, A Meta-Analysis of Hypothetical Bias in Stated Preference Valuation, Environmental and Resource Economics, 2005. 30: 313–325.

Murray, C., B. Sohngen, and L. Pendleton, Valuing Water Quality Advisories and Beach Amenities in the Great Lakes, Water Resources Research, 2001. 37: 2583-2590.

Nelson, J. P., Residential Choice, Hedonic Prices, and the Demand for Urban Air Quality, Journal of Urban Economics, 1978. 5: 357-369.

Nelson, J. P., Highway Noise and Property Values, Journal of Transport Economics and Policy, 1982. 16: 117-138.

Netusil, N.R., The Effect of Environmental Zoning and Amenities on Property Values: Portland, Oregon, Land Economics, 2005. 81: 227-246.

Noll, H., European Survey Data: Rich Sources for Quality of Life Research. In: V. Møller et al. (eds.), Barometers of Quality of Life Around the Globe, Springer, Netherlands, 2008.

Nourse, H.O., The Effect of Air Pollution on House Values, Land Economics, 1967. 43: 181-189.

O'Byrne, P. H., J. P. Nelson and J. J. Seneca, Housing Values, Census Estimates, Disequilibrium, and the Environmental Cost of Airport Noise: A Case Study of Atlanta, Journal of Environmental Economics and Management, 1985. 12: 169-178.

Olsen, D., J. Richards and D. Scott, Existence and sport values for doubling the size of Columbia river basin salmon and steelhead runs, Rivers, 1991. 2: 44–56.

Pacific Fisheries Information Network (PacFIN) retrieval dated 7 July 2009, Pacific States Marine Fisheries Commission, Portland, Oregon (www.psmfc.org).

Pacific Fishery Management Council. Review of 2009 Ocean Salmon Fisheries. Pacific Fishery Management Council, Portland, Oregon, 2010.

Peters C., A. Gentry, and R. Mendelsohn, Valuation of an Amazonian rain forest, Nature, 1989. 339: 655–656.

Pollnac, R.B., S. Abbott-Jamieson, C. Smith, M.L. Miller, P.M. Clay, and B. Oles, Toward a Model for Fisheries Social Impact Assessment, Marine Fisheries Review, 2006. 68: 1-18.

Poor, P.J., K.L. Pessagno, and R.W. Paul, Exploring the Hedonic Value of Ambient Water Quality: A Local Watershed-Based Study, Ecological Economics, 2007. 60: 797-806.

Puget Sound Partnership, Puget Sound Action Agenda, Protecting and Restoring the Puget Sound Ecosystem by 2020, Olympia, WA, 2008.

Puget Sound Partnership, Ecosystem Status & Trends: A 2009 Supplement to State of the Sound Reporting, Olympia, WA, 2009a.

Puget Sound Partnership, Identification of Ecosystem Components and Their Indicators and Targets, Technical Memorandum, Olympia, WA, 2009b.

Puget Sound Partnership, Identification, Definition and Rating of Threats to the Recovery of Puget Sound, Technical Memorandum, Olympia, WA, 2009c.

Puget Sound Partnership, Using Results Chains to Develop Objectives and Performance Measures for the 2008 Action Agenda, Technical Memorandum, Olympia, WA, 2009d.

Read, D., Utility theory from Jeremy Bentham to Daniel Kahneman. Department of Operational Research, London School of Economics, Working Paper No: LSEOR 04-64, 2004.

Reaves, D.W., R.A. Kramer, and T.P. Holmes, Valuing the endangered red cockaded woodpecker and its habitat: a comparison of contingent valuation elicitation techniques and a test for embedding. AAEA meetings paper. 1994.

Redman, C.L., J.M. Grove, and L.H. Kuby. Integrating Social Science into the Long-Term Ecological Research (LTER) Network: Social Dimensions of Ecological Change and Ecological Dimensions of Social Change. Ecosystems, 2004. 7: 161-171.

Rehdanz, K. and D. Maddison, Climate and happiness, Ecological Economics, 2005. 52: 111–125.

Reichert, A.K., Impact of a Toxic Waste Superfund Site on Property Values, The Appraisal Journal, 1997. 65: 381-392.

Reichert, A.K., M. Small, and S. Mohanty, The Impact of Landfills on Residential Property Values, The Journal of Real Estate Research, 1992. 7: 297-314.

Richardson, L. and J. Loomis, The total economic value of threatened, endangered and rare species: An updated meta-analysis, Ecological Economics, 2009. 68: 1535-48.

Rubin, J., G. Helfand, and J. Loomis, A benefit–cost analysis of the northern spotted owl, Journal of Forestry, 1991. 89: 25–30.

Samples, K., and J. Hollyer, Contingent valuation of wildlife resources in the presence of substitutes and complements. In: Johnson, R., Johnson, G. (Eds.), Economic Valuation of Natural Resources: Issues, Theory and Application. Westview Press, Boulder, CO, 1989.

Schmidt, L., and P.N. Courant, Sometimes Close is Good Enough: The Value of Nearby Environmental Amenities, Journal of Regional Science, 2006. 46: 931–951.

Sharpe, A., A survey of indicators of economic and social well-being. Background paper prepared for Canadian Policy Research Networks. Ottawa, Ontario, 1999.

Slesnick, D., Empirical Approaches to the Measurement of Welfare, Journal of Economic Literature, 1998. 36: 2108–2165.

Smith, R.H., E. Diener, and D.H. Wedell, Intrapersonal and Social Comparison Determinants of Happiness: A Range- Frequency Analysis, Journal of Personality and Social Psychology, 1989. 56: 317–25.

Smith, V.K., and W. H. Desvousges, The Value of Avoiding a LULU: Hazardous Waste Disposal Sites, Review of Economics and Statistics, 1986. 68: 293-299.

Smolen, G.E., G. Moore, and L.V. Conway, Economic Effects of Hazardous Chemical and Proposed Radioactive Waste Landfills on Surrounding Real Estate Values, The Journal of Real Estate Research, 1992. 7: 283-295.

Solberg, E.C., E. Diener, D. Wirtz, R.E. Lucas, and S. Oishi, Wanting, Having, and Satisfaction: Examining the Role of Desire Discrepancies in Satisfaction with Income, Journal of Personality and Social Psychology, 2002. 83: 725-734.

Stanley, D.L., Local perception of public goods: recent assessments of willingness-to-pay for endangered species, Contemporary Economic Policy, 2005. 2: 165–179.

State of Washington, 60th Legislature, Engrossed Substitute Senate Bill 5372, as Amended by the House, 2007.

Stevens T., J. Echeverria, R. Glass, T. Hager, and T. More, Measuring the existence value of wildlife: what do CVM estimates really show?, Land Economics, 1991. 67: 390–400.

Stiglitz, J.E., S. Amartya, and J. Fitoussi, The measurement of economic performance and social progress revisited, Documents de Travail de l'OFCE 2009-33, Observatoire Francais des Conjonctures Economiques (OFCE), 2009

Stinson, D.W., J.W. Watson, and K.R. McAllister, Washington State Status Report for the Bald Eagle, Washington Department of Fish and Wildlife, Olympia. 2007. 86 + viii pp.

Swanson, C.S., Economics of Endangered Species: Bald Eagles on the Skagit River Bald Eagle Natural Area, Washington, Transactions of the 61st North American Wildlife and Natural Resources Conference, 1996. 61: 293-300.

Taylor, S. M., B. E. Breston, and F. L. Hall, The Effect of Road Traffic Noise On House Prices, Journal of Sound and Vibration, 1982. 80: 523-541.

Thayer, M., H., Albers and M. Rahmatian, The Benefits of Reducing Exposure to Waste Disposal Sites: A Hedonic Housing Value Approach, Journal of Real Estate Research, 1992. 7: 265-282.

Tseng, F., and Y. Chiu, Hierarchical fuzzy integral stated preference method for Taiwan's broadband service market, Omega, 2005. 33: 55-64.

Tversky, A., and D. Griffin, Endowment and Contrast in Judgments of Well- Being, in Strategy and Choice. Richard J. Zeckhauser, ed. Cambridge and London: MIT Press, 1991, pp. 297–318.

van Praag, B.M.S., and B,.E. Baarsma, Using happiness surveys to value intangibles: The case of airport noise, Economic Journal, 2005. 115: 224–246.

Vera-Toscano E., V. Ateca-Amestoy, and R. Serrano-del-Rosal, Building Financial Satisfaction, Social Indicators Research, 2006. 77: 211-243.

Viscusi, W.K., J. Huber, and J. Bell, The Economic Value of Water Quality, Environmental Resources and Economics, 2008. 41:169–187.

Vitarelli, A., Happiness Metrics in Federal Rulemaking. Yale Journal on Regulation, 2010. 27: 115-158.

Walker, B., S. Carpenter, J. Anderies, N. Abel, G. S. Cumming, M. Janssen, L. Lebel, J. Norberg, G. D. Peterson, and R. Pritchard, Resilience management in social-ecological systems: a working hypothesis for a participatory approach, Conservation Ecology, 2002. 6(1): 14.

Warr, P., Well-being and the Workplace. In D. Kahneman, E. Diener and N. Schwarz (eds.). Well-Being: The Foundations of Hedonic Psychology, New York: Russel Sage Foundation, 1999, pp. 392-412.

Welsch, H., Preferences over prosperity and pollution: environmental valuation based on happiness surveys, Kyklos, 2002. 55: 473–494.

Welsch, H., Environment and happiness: valuation of air pollution using life satisfaction data, Ecological Economics, 2006. 58: 801–813.

Welsch, H., and J. Kühling, Using Happiness Data For Environmental Valuation: Issues And Applications, Journal of Economic Surveys, 2009. 23: 385–406.

Winkelmann, L., and R. Winkelmann, Why are the unemployed so unhappy? Economica, 1998. 65: 1–15.

Yohe, G.W.,, asnd M.E. Schlesinger, Sea-level change: the expected economic cost of protection or abandonment in the United States. Clim.ate Change, 1998. 38:447–472.

Zabel, J.E., and K.A. Kiel, K.A., Estimating the demand for air quality in four US cities, Land Economics, 2000. 76: 174–194.

Zerbe, R.O., and A.S. Bellas, A Primer for Benefit-Cost Analysis. Northampton, MA: Edward Elgar Publishing, Inc., 2006.

Zerbe, R. O., and D. Dively, Benefit-Cost Analysis in Theory and Practice. New York: HarperCollins College Publishers, 1994.

Figures

Table 1. Examples of studies using the direct, subjective measurement approach for estimating links between HWB and environmental quality	
Climate	Becchetti et al. (2007) Frijters and van Praag (1998) Rehdanz and Maddison (2005)
Droughts	Carroll et al. (2009)
Air pollution	Welsch (2002) Welsch (2006) Di Tella and MacCulloch (2007) Luechinger (2009)
Airport noise nuisance	van Praag and Baarsma (2005)
Flood hazards	Luechinger and Raschky (2009
Water pollution	Israel and Levinson (2003)

Table 2. Examples of market landings and revenues in 2008 for species harvested in Puget Sound (Pacific States Marine Fisheries Commission, 2009)

Species common name	Aquaculture		Commercial (Non-tribal)		Commercial (tribal)	
	Landed weight (lbs)	Landed revenue ($)	Landed weight (lbs)	Landed revenue ($)	Landed weight (lbs)	Landed revenue ($)
Geoduck	4,122,429	$25,353,623	2,290,914	$6,188,422	3,197,846	$11,759,146
Chum Salmon			4,196,843	$3,552,228	4,689,451	$3,717,760
Manila Clam	7,149,458	$18,385,757	5,690	$10,811	788,595	$1,268,706
Dungeness Crab			2,837,020	$6,785,143	4,013,664	$10,198,513
Blue or Bay Mussel	2,963,216	$5,293,124			400	$600
Pacific Oyster	2,222,221	$7,498,498	21,238	$84,094	388,746	$1,253,802
Coho Salmon			205,236	$289,293	1,966,139	$3,389,613
Chinook Salmon			180,821	$566,347	1,387,001	$3,613,382

Table 3. Examples of studies using the hedonic approach for estimating links between HWB and environmental quality	
Air pollution	Anderson and Crocker (1971) Chattopadhyay (1999) Freeman III (1974) Graves et al. (1988) Harrison Jr. and Rubinfeld (1978) Murdoch and Thayer (1988) Nelson (1978) Nourse (1967), Zabel and Kiel (2000)
Water quality	Boyle et al., (1999) Leggett and Bockstael (2000) Poor et al. (2006) Epp and Al-Ani (1979) Gibbs et al. (2002) Halstead et al. (2003)
Noise	Hall et al. (1978) Nelson, J. P. (1982) O'Byrne et al. (1985) Taylor et al. (1982)
Solid waste sites	Havlicek et al. (1971) Reichert et al. (1992) Thayer and Rahmatian (1992)
Shore erosion protection	Kriesel et al. (1993)
Toxic waste sites	Kiel, K.A. (1995) Kohlhase (1991) Reichert (1997) Smith and Desvousges (1986) Smolen et al. (1992)

Table 4. Average recreation values based on studies from Oregon and Washington that use the revealed preference approach (Loomis, 2005)

Activity	Value per day ($2004)	Number of studies
Fishing	$41.98	5
Hiking	$23.98	5
Hunting	$35.27	5
Motorboating	$12.48	1
Swimming	$6.06	1
Wildlife viewing	$35.00	3

Table 5. Examples of studies that use the stated preference approach to estimate the economic non-use value of a species (Richardson and Loomis, 2008)

Arctic grayling	Duffield and Patterson (1992)
Atlantic salmon	Stevens et al. (1991)
Bald eagle	Boyle and Bishop (1987) Stevens et al. (1991) Swanson (1996)
Bighorn sheep	King et al. (1988)
Blue whale	Hageman (1985)
Bottlenose dolphin	Hageman (1985)
Gray whale	Hageman (1985) Loomis and Larson (1994)
Gray wolf	Duffield (1991, 1992) Duffield et al. (1993) Chambers and Whitehead (2003)
Humpback whale	Samples and Hollyer (1989)
Mexican spotted owl	Loomis and Ekstrand (1997) Giraud et al. (1999)
Monk seal	Samples and Hollyer (1986)
Northern spotted owl	Rubin et al. (1991) Hagen et al. (1992)
Northern elephant seal	Hageman (1985)
Peregrine falcon	Kotchen and Reiling (2000)
Red-cockaded woodpecker	Reaves et al. (1994)
Riverside fairy shrimp	Stanley (2005)
Salmon	Olsen et al. (1991) Loomis (1996) Layton et al. (2001) Bell et al. (2003)
Sea otter	Hageman (1985)
Silvery minnow	Berrens et al. (1996)
Squawfish	Cummings et al. (1994)
Steller sea lion	Giraud et al. (2002)
Striped shiner	Boyle and Bishop (1987)
Whooping crane	Bowker and Stoll (1988)
Wild Turkey	Stevens et al. (1991)

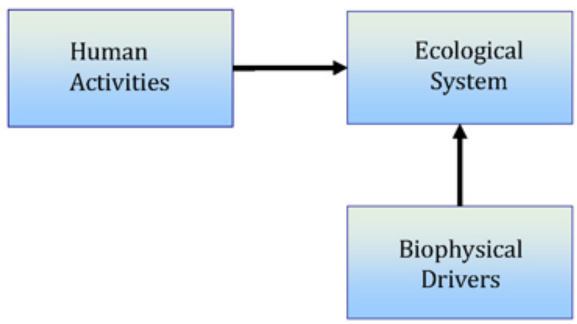

Figure 1: A simplistic view of an ecological system that is affected by but separate from human systems. Adapted from Redman, *et al.* (2004).

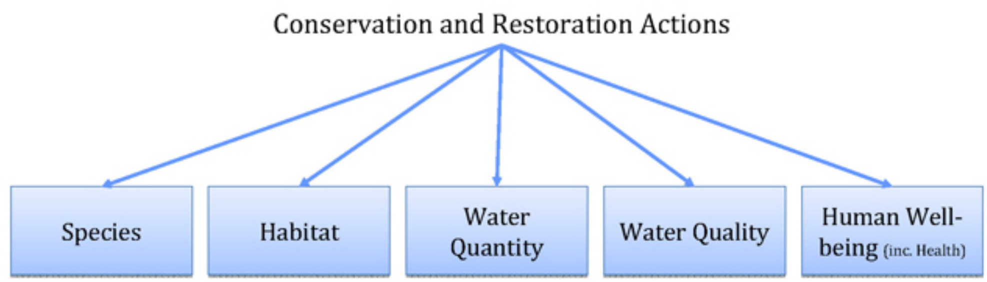

Figure 2: A framework for evaluating the effects of conservation and restoration actions on the Partnership's goals, where each goal is considered with a distinct set of metrics.

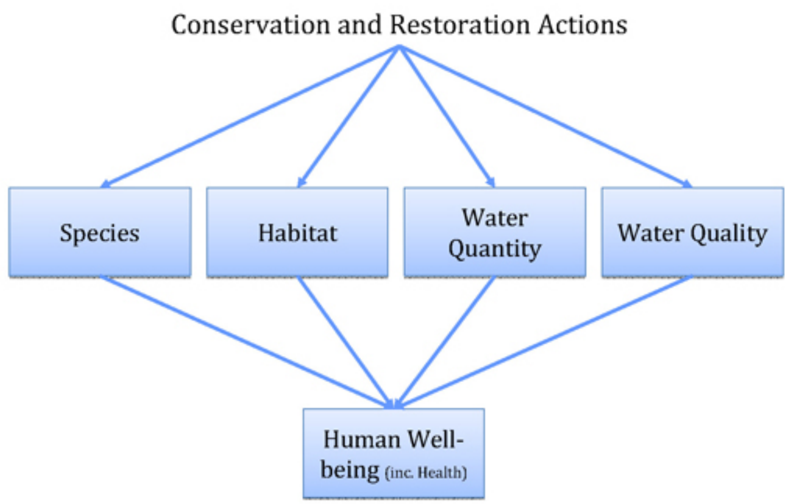

Figure 3: A framework for evaluating the effects of conservation and restoration actions on the Partnership's ecological goals using human well-being as a common metric.

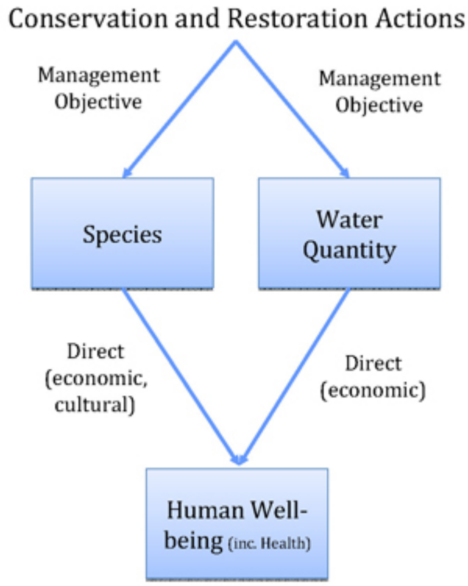

Figure 4: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that are directed at species and water quality.

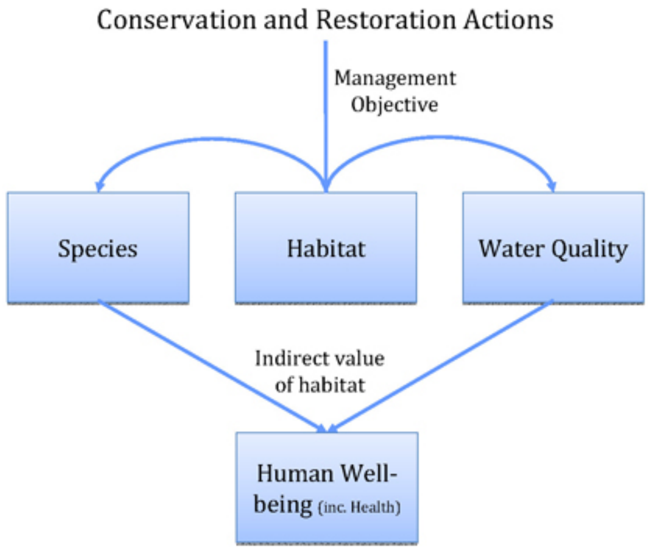

Figure 5: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that are directed at habitat, which in turn affects species and water quality and thereby affecting human well-being.

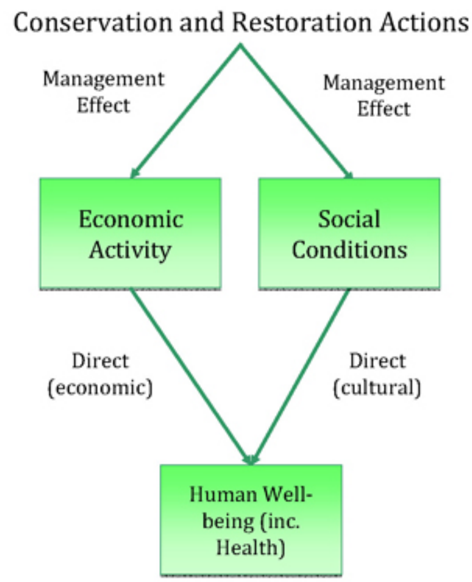

Figure 6: An example of how human well-being can be used as a metric to assess the effects of conservation and restoration actions that affect important components that support human well-being but which are not the objectives of management.

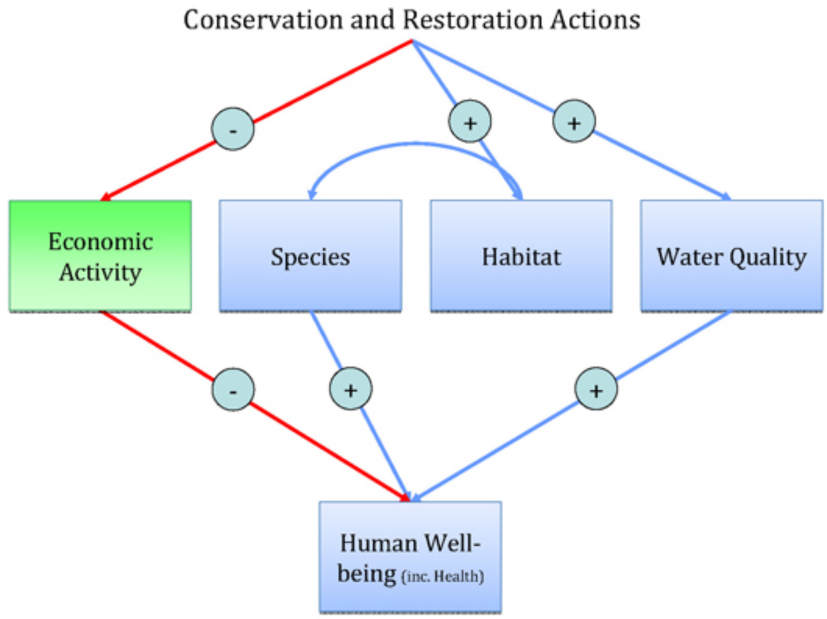

Figure 7: An example of how human well-being can be affected by conservation and restoration actions along multiple pathways.

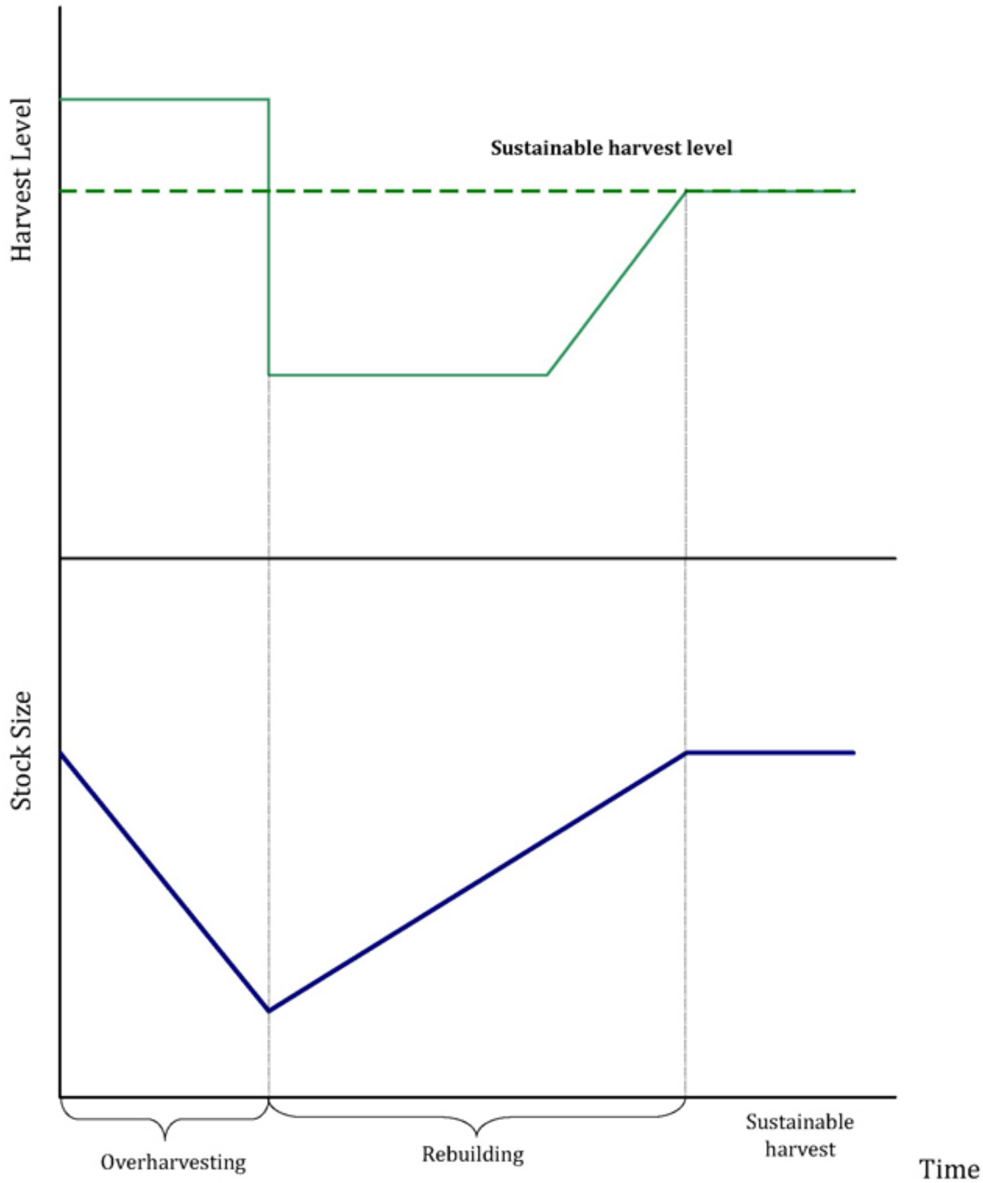

Figure 8 (upper panel): After a period of exceeding the sustainable level, harvest is reduced to allow the fish population to rebuild, and then is gradually increased to its sustainable level. These changes are mirrored by changes in current HWB.
Figure 8 (lower panel): Changes in the stock size accurately track potential future harvests and so can be used as an indicator for future HWB.

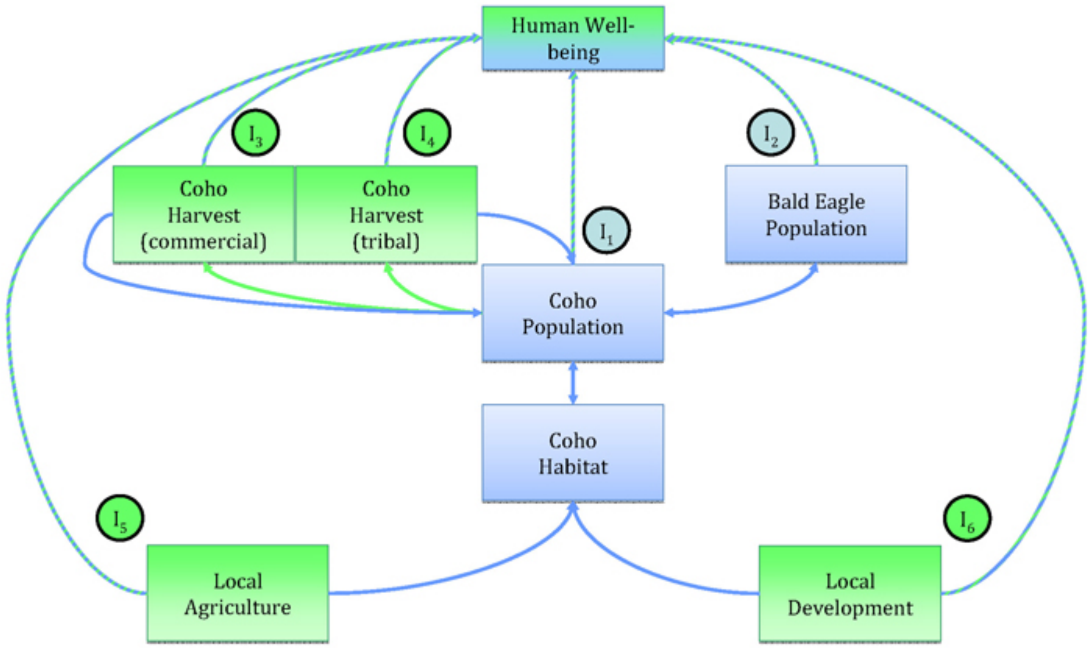

Figure 9: An example of the connections between biophysical and human-based components of the Puget Sound ecosystem, and between those components and human well-being. Identifying these connections can facilitate the identification and evaluation of biophysical and human well-being indicators.

Chapter 2A. The Biophysical Condition of Puget Sound

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Section 1. Introduction

Our objective in this section is to review the status and trends of biophysical components of Puget Sound that speak to the Puget Sound Partnerships key goals: species and food webs, habitats, water quality and water quantity. Each of these goals are multi-facetted, and a nearly limitless range of topics could be covered. Indeed, one of the qualities that make Puget Sound a natural treasure is the diversity of species and habitats that it supports. This diversity precludes detailed treatment of all ecosystem components and requires thoughtful selection of metrics that speak to ecological condition and policy goals.

An ideal process for selecting components would be a sequential approach allowing us to use the framework developed in Chapter 1 to evaluate multiple indicators followed by an analysis of data availability, status and trends therein. However, time constraints required that we work in parallel with the Chapter 1 effort, so our choice of focal components and our reporting is largely independent of that process. We do not use the term "indicators" when referring to these components because they have not been formally vetted as such.

Lacking a formal procedure or framework to select focal biophysical components, we adopted two overarching considerations in selecting components: metrics should be ecologically or policy relevant attributes of Puget Sound, and must have been the focus of sufficient study to permit status evaluation. Consequently, species that are recognized as important in the Puget Sound ecosystem, but for which sufficient data do not exist, were excluded from this analysis. Omissions based on data insufficiencies can be used to help guide decisions regarding data collection programs in the future. Additional guiding principles and considerations included the following: 1) culturally important species for which there are clear policy goals (e.g., harvested species, iconic species such as killer whales) were included whenever possible, along with critical species and habitats upon which they rely; 2) species of particular conservation concern were incorporated; 3) water quality and water quality components were chosen to reflect the topical emphasis of scientific study in each of those disciplines; 4) species that have been specifically identified as ecosystem indicators (via peer reviewed publications) were considered whenever possible.

This set of principles provided criteria that allowed a systematic approach to selection of components to include in this analysis. However, it did result in some noteworthy exclusions. For example, the status and trends of invasive species (e.g., *Spartina*, *Ciona*) are not reported. Analysis of zooplankton community composition and trends is limited by the paucity of data. Ocean acidification, a growing concern with potentially substantial impacts on shellfish aquaculture and natural communities, is not treated here. These and other omissions are not intended to imply that these are not important issues or components of the Puget Sound ecosystem, and we anticipate that the next iteration of the Puget Sound Science update can consider a broader range of metrics.

The ecosystem components treated in this chapter clearly emphasize marine and freshwater elements of the Puget Sound Watershed. This emphasis reflects the historical focus of the Puget Sound Science Update and the specific expertise of the lead authors. Even so, we selected terrestrial topics that have some linkage to aquatic portions of the watershed. We anticipate that future iterations of the Puget Sound Science Update will take a broader view and include many more terrestrial topics than we could incorporate in the present document.

There is a growing need for ecosystem assessments to guide ecosystem-based management. While the present evaluation might be considered a contribution to such an assessment, it is not an ecosystem assessment per se. Instead, it is an assessment of several ecosystem components. A full ecosystem assessment would also include a conceptual framework that links biological, physical and chemical processes and reports on key drivers and responses of each. Moreover, a quantitative synthesis of status and trends across all ecological and policy-relevant attributes of Puget Sound will provide a substantial advance.

Throughout, we aimed to vet available information to include only those results and conclusions that had undergone prior review. We recognized in advance that maintaining a requirement of peer-reviewed publication in scientific journals would be inappropriate: much of the scientific work on Puget Sound derives from long term monitoring that is not published in such journals. We therefore considered agency documents that were part of research reporting series to be sufficiently reviewed to be included in this chapter. This process revealed considerable differences among local agencies in the transparency of review processes for reports. There is a need for consistent standards and reporting practices among these agencies to permit an assessment of the thoroughness of reviews. We generally avoided citing previous iterations of the Puget Sound Science update as primary sources, because the nature and extent of review of components of those documents is also not clear. In some cases, monitoring data were used directly provided that the procedures used in collecting them had been reviewed and published.

Given these constraints, this chapter is not intended to be the final word on indicators for evaluating the status of Puget Sound. Indeed, Chapter 1 of the 2010 Puget Sound Science Update provides a substantial advance in improving the capacity to select ecologically meaningful indicators. Future versions of the Puget Sound Science Update will clearly benefit from the foundation that the present effort provides.

This chapter is organized primarily along the four Puget Sound Partnership goals, with separate sections for each ecosystem component. Within each summary, we provide background and rationale for inclusion in the Chapter, a brief treatment of threats and drivers to give the needed context. More thorough treatment of threats and drivers is provided in Chapter 3. We include in each section a synthesis of key data gaps and uncertainties. In some cases the uncertainties are scientific: uncertainties that can be resolved through additional scientific study. In other cases the uncertainties reflect emerging concepts, hypotheses and explanations that have not yet been vetted through a formal review process.

Species and Food Webs

1. Bivalves

Background

Molluscs in the Class Bivalvia feed on phytoplankton and detrital particles suspended in the water column, serving as a key trophic link between microscopic primary producers and higher consumers. Epibenthic bivalves can function as ecosystem engineers through the provision of hard substrate and three-dimensional biogenic structure, while infaunal bivalves can function as engineers through physical alteration of soft substrate habitats. Numerous native and non-native species of bivalves occur in Puget Sound, including important aquaculture species such as Pacific oysters (*Crassostrea gigas*), non-native invasive species such as the purple varnish clam (*Nutallia obscurata*), and species targeted in recreational fisheries (e.g., native littleneck clams and non-native Manila clams). The native geoduck clam, *Panopea generosa*, is valued as a commercially-fished species and as an aquaculture species. The native Olympia oyster, *Ostrea lurida*(also known as *Ostreola conchaphila*) currently is a restoration target in Puget Sound, having been depleted through human activities in the last century.

Geoduck clams

Geoducks are large Hiatellid clams distributed from Alaska to California. They can grow to shell lengths of 20 cm (Bureau et al. 2002), and are characterized by large fleshy siphons that can reach lengths of 1m. Geoducks are broadcast spawners with larval periods of 16 - 47 days (Goodwin and Pease 1989). After settlement, they exhibit limited mobility for 2-4 weeks, then burrow into the sand and begin feeding. Individuals are thought to reach maximum size within the first 10 years of life (Goodwin and Pease 1989), and can live for up to 168 years. Their longevity could render them particularly susceptible to over-exploitation (Orensanz et al. 2004).

In Puget Sound, geoducks occur primarily in low intertidal and subtidal habitats and are most abundant at depths of up to 20m, although observations of deeper individuals have been reported (Goodwin and Pease 1989). Found primarily in soft sediments consisting of sand and sand-mud, geoducks are contagiously distributed throughout the major basins of Puget Sound (Goodwin and Pease 1990). In a survey of 8,589 SCUBA transects, Goodwin and Pease (1990) found that geoduck abundance ranged from densities of 0 to 22.5 individuals/m2, with an average density of 1.7 individuals/m2. They found the highest densities in southern Puget Sound and in Hood Canal (Goodwin and Pease 1990).

Recreational and commercial fisheries for geoduck exist in Puget Sound. The recreational fishery typically occurs in intertidal habitats, while the commercial fishery occurs in subtidal habitats in areas leased from the State of Washington. Because the fishery is prosecuted in leased tracts, it is jointly managed by the Washington State Department of Natural Resources (WDNR) and the Washington Department of Fish and Game (WDFW). The current target for the commercial fishery in Puget Sound is 2.7% of the exploitable biomass based on a static value of 40% of the Maximum Sustainable Yield (MSY) (Bradbury et al. 2000). Recruitment of geoducks appears to be highly variable and driven by climatic forcing (Orensanz et al. 2004, Valero et al. 2004). Based on the combination of highly variable recruitment and long life span, Orensanz et al. (2004) caution that static exploitation targets may not be appropriate for this species. Geoduck abundance in Puget Sound is augmented through aquaculture, the ecological effects of which are not well understood (Feldmann et al. 2004, Straus et al. 2008).

Olympia oyster

As ecosystem engineers, oysters play an important role in the populations, communities and food webs where they occur (reviewed in Ruesink et al. 2005). Oyster beds provide structure and biogenic habitat for a suite of other invertebrates and fish (e.g., Lenihan et al. 2001). They also modify the physical and chemical properties of ambient water through feeding and excretion, maintaining high water clarity and conditions beneficial to macrophytes (Jackson et al. 2001, Ruesink et al. 2005).

The native Olympia oyster occurs from Alaska to Baja California, Mexico (Polson and Zacherl 2009). The size of the particles or phytoplankton ingested by oysters is determined by the size of their gills. Olympia oysters have larger gills and thus likely ingest larger particles than the common non-native Pacific oyster *Crassostrea gigas* (Couch and Hassler 1989). Olympia oysters are preyed upon by birds such as sea ducks and by crabs (Couch and Hassler 1989). They are relatively small, rarely reaching sizes greater than 5 cm, and have slow growth rates, typically reaching maturity after 4 years (Baker 1995, White et al. 2009b). Unlike many bivalves, fertilization is internal and larvae brood for 10-12 days within the mantle of females before spending 11-16 days as planktonic larvae (Dethier 2006). Olympia oyster spat have fairly narrow requirements for settlement, preferring hard, rugose substrates such as adult oyster shells (Trimble et al. 2009, White et al. 2009b). Beds of Olympia oysters are typically subtidal and individuals are known to be sensitive to extremes in temperature and desiccation stress (e.g., Baker 1995).

Status and Trends

Geoduck

Geoduck abundances for individual tracts throughout Puget Sound are estimated based on diver surveys conducted by WDFW according the methods described in Bradbury et al. (2000) and are posted online as part of the Geoduck Atlas , but abundances at the basin or sound-wide scales have not been summarized or published. Similarly, published fishery-independent population abundance data on trends in geoduck abundances are lacking.

Olympia Oyster

Olympia oysters in Washington state have been heavily exploited (Kirby 2004) and currently exist at abundances far lower than were reported historically (White et al. 2009a) (Figure 1). In Puget Sound, abundance was greatly reduced in the early 1900s despite the implementation of reserves throughout the Sound. Industrial pollution from paper mills is thought is thought to have contributed to the lack of effectiveness of the reserves (White et al. 2009a). The continued lack of population recovery is thought to be driven by a combination of limitations in the amount of preferred settlement substrate (adult conspecifics), competition with non-native oysters, and predation from introduced predators such as the Japanese drill *Ocinebrina inornata* (Buhle and Ruesink 2009, Trimble et al. 2009, White et al. 2009b). Their sensitivity to environmental extremes further restricts the habitats they can occupy (Trimble et al. 2009). Because of their low abundance, Olympia oysters currently are listed as a Washington State Candidate Species by WDFW . A number of projects for restoration of Olympia oyster populations have been initiated in Puget Sound (e.g., Brumbaugh and Coen 2009, Dinnel et al. 2009, White et al. 2009b).

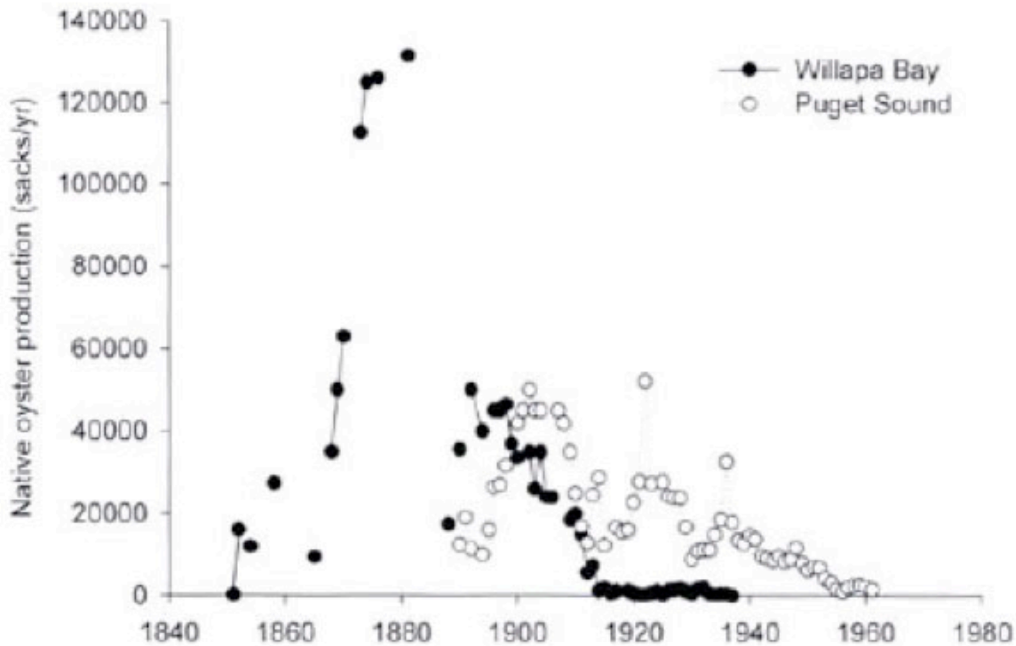


Figure 1. Olympia oyster harvest (1 sack is equal to approximately 4,000 individuals) in Willapa Bay (filled circles) and Puget Sound (open circles) from the mid 19th to mid 20th century based on Washington Marine Fish and Shellfish Landings (figure from White et al. 2009) (reprinted with permission from the Journal of Shellfish Research).

UNcertainties

There are several aspects of the current understanding of geoduck and Olympia oyster populations that are lacking. Geoduck tracts are surveyed frequently by WDFW yet estimates of basin and Sound-wide population status or trends have not been conducted. As such, spatial and temporal trends in geoduck abundances are not known for Puget Sound. Further, while cultivation of geoducks augments population abundances, the ecological effects of geoduck aquaculture practices in Puget Sound are not well understood (Feldmann et al. 2004, Straus et al. 2008). The sensitivity of Olympia oyster populations to abiotic stress and to predation from non-native predators pose challenges to the undertaking of restoring them to their former abundances and such the outcome of such efforts remains uncertain.

Summary

Native bivalves are essential components of the Puget Sound ecosystem. Geoduck clams are extremely long-lived, rendering them potentially susceptible to overexploitation. While geoduck abundance is estimated at small scales (tracts), published accounts of Sound-wide estimates of population status and trends are lacking. Abundances of Olympia oysters have been very low in Puget Sound since the 1940s, despite the fact that they are no longer targeted by fisheries. The importance of native oysters to ecosystems has prompted restoration efforts throughout Puget Sound.

Literature Cited

Baker, P. 1995. Review of ecology and fishery of the Olympia oyster, *Ostrea lurida*, with annotated bibliography. Journal of Shellfish Research 14:501-518.

Bradbury, A., B. Sizemore, D. Rothaus, and M. Ulrich. 2000. Stock assessment of subtidal geoduck clams (*Panopea abrupta*) in Washington. Washington Department of Fish and Wildlife, Olympia, WA.

Brumbaugh, R. D., and L. D. Coen. 2009. Contemporary approaches for small-scale oyster reef restoration to address substrate versus recruitment limitation: a review and comments relevant for the Olympia oyster, *Ostrea lurida* Carpenter 1864.(Report). Journal of Shellfish Research 28:147(115).

Buhle, E. R., and J. L. Ruesink. 2009. Impacts of invasive oyster drills on Olympia oyster (*Ostrea lurida* Carpenter 1864) recovery in Willapa Bay, Washington, United States.(Report). Journal of Shellfish Research 28:87(10).

Bureau, D., W. Hajas, N. W. Surry, C. M. Hand, G. Dovey, and A. Campbell. 2002. Age, size structure and growth parameters of geoducks (*Panopea abrupta* Conrad 1849) from 34 locations in British Columbia sampled between 1993 and 2000. Canadian Technical Report of Fisheries and Aquatic Sciences, Canada.

Couch, D., and T. J. Hassler. 1989. Species profiles: Life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest): Olympia Oyster. United States Fish and Wildlife Service Biological Report.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound.Puget Sound Nearshore Partnership Report No. 2006-04, Seattle, WA.

Dinnel, P. A., B. Peabody, and T. Peter-Contesse. 2009. Rebuilding Olympia oysters, *Ostrea lurida* Carpenter 1864, in Fidalgo Bay, Washington.(Report). Journal of Shellfish Research 28:79(77).

Feldmann, K., B. Vadopalas, D. Armstrong, C. Friedman, R. Hilborn, K. Naish, J. Orensanz, J. Ruesink, A. Suhrbier, A. Christy, D. Cheney, and J. Davis. 2004. Comprehensive literature review and synopsis of issues related to geoduck (*Panopea abrupta*) ecology and aquaculture production. Prepared for the Washington State Department of Natural Resources, Olympia, WA.

Goodwin, C. L., and B. C. Pease. 1990. Geoduck, Panopea abrupta (Conrad, 1849), size, density, and quality as related to various environmental parameters in Puget Sound, Washington. Journal of Shellfish Research 10:65-77.

Goodwin, L., and B. Pease. 1989. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (Pacific Northwest) -- Pacific geoduck clam. . United States Fish and Wildlife Service Biological Report, Brinnon, WA.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. Science 293:629-637.

Kirby, M. X. 2004. Fishing down the coast: Historical expansion and collapse of oyster fisheries along continental margins. Proceedings of the National Academy of Sciences of the United States of America 101:13096-13099.

Lenihan, H. S., C. H. Peterson, J. E. Byers, J. H. Grabowski, G. W. Thayer, and D. R. Colby. 2001. Cascading of habitat degredation: Oyster reefs invaded by refugee fishes escaping stress. Ecological Applications 11:764-782.

Orensanz, J. M., C. M. Hand, A. M. Parma, J. Valero, and R. Hilborn. 2004. Precaution in the harvest of Methuselah's clams - the difficulty of getting timely feedback from slow-paced dynamics. Canadian Journal of Fisheries and Aquatic Sciences 61:1355-1372.

Polson, M. P., and D. C. Zacherl. 2009. Geographic distribution and intertidal population status for the Olympia oyster, *Ostrea lurida* Carpenter 1864, from Alaska to Baja.(Report). Journal of Shellfish Research 28:69(69).

Ruesink, J. L., H. S. Lenihan, A. C. Trimble, K. W. Heiman, F. Micheli, J. E. Byers, and M. C. Kay. 2005. Introduction of non-native oysters: Ecosystem effects and restoration implications. Annual Review of Ecology, Evolution, and Systematics 36:643-689.

Straus, K. M., L. M. Crosson, and B. V. Vadopalas. 2008. Effects of geoduck aquaculture on th environment: A synthesis of current knowledge. Washington Sea Grant Technical Report, Seattle, WA.

Trimble, A. C., J. L. Ruesink, and B. R. Dumbauld. 2009. Factors preventing the recovery of a historically overexploited shellfish species, *Ostrea lurida* Carpenter 1864.(Report). Journal of Shellfish Research 28:97(10).

Valero, J. L., C. Hand, J. M. Orensanz, A. M. Parma, D. Armstrong, and R. Hilborn. 2004. Geoduck (*Panopea abrupta*) recruitment in the Pacific Northwest: Long-term changes in relation to climate. California Cooperative Oceanic Fisheries Investigations Reports 45:80-86.

White, J., J. L. Ruesink, and A. C. Trimble. 2009a. The nearly forgotten oyster: *Ostrea lurida* Carpenter 1864 (Olympia oyster) history and management in Washington State.(Report). Journal of Shellfish Research 28:43(47).

White, J. M., E. R. Buhle, J. L. Ruesink, and A. C. Trimble. 2009b. Evaluation of Olympia oyster (*Ostrea lurida* carpenter 1864) status and restoration techniques in Puget sound, Washington, United States.(Report). Journal of Shellfish Research 28:107(106).

Pinto abalone

Background

Pinto abalone (*Haliotis kamtschatkana*) were once widely distributed throughout the waters of British Columbia and Washington state. In recent decades, populations have undergone sharp declines, likely in response to the combined stressors of overharvest, poaching, and sub-optimal environmental conditions (Campell 2000). Known for their large, muscular foot and their pearlescent oval shell, pinto abalone are slow-growing, long-lived marine snails and are typically found in nearshore rocky habitats in semi-exposed or exposed coastal regions. More than 60 abalone species are found worldwide but the pinto, or northern, abalone is the only species found in Washington State, where they range from Admiralty Inlet to the San Juan Islands and the Strait of Juan de Fuca and are typically found at depths to about 20 m (Bouma 2007).

Abalone are important herbivores in nearshore habitats, feeding primarily on drift macroalgae such as kelp and benthic diatom films. They can structure subtidal communities through the maintenance of substrata dominated by crustose coralline algae and through the facilitation of conspecific settlement. The larvae are planktonic and settle after approximately 7 -10 days in response to cues from both crustose coralline algae and from adults. Juvenile pinto abalone are cryptic until they reach a shell length of >50 mm.

Abalone are broadcast spawners. Consequently, the number and proximity of spawning adults determines the likelihood of successful fertilization (e.g., Babcock and Keesing 1999, Miner et al. 2006). At low population numbers, fertilization success may be low or nil, potentially limiting population recovery from overharvesting (Rothaus et al. 2008).

Status

The Washington Department of Fish and Wildlife (WDFW) regularly monitors the abundance of pinto abalone at 10 index stations throughout the San Juan Archipelago (Rothaus et al. 2008) (Figure 1). Because pinto abalone are highly patchy, cryptic and frequently associate with microhabitats such as rock crevices or patches of coralline algae that may themselves be patchily distributed, total abundances are not measured (Rothaus et al. 2008). Rather, repeated surveys at a system of index sites are conducted so that temporal trends in abalone abundance may be detected. The WDFW sites are composed mostly of bedrock and boulders encrusted with coralline algae, and support assemblages of kelp and other macroalgae (Rothaus et al. 2008). The sites range in size from 135 m2 to 380 m2, and individual animals are counted and measured during each survey.

Data from surveys made in 2006 showed an overall mean abalone density of 0.04 m-2 (Rothaus et al. 2008), which is well below the minimum densities for successful reproduction (0.15 individuals m-2) and recruitment (1 individual m-2) reported respectively by Babcock and Keesing (1999) and by Miner et al. (2006) for congeners of the pinto abalone.

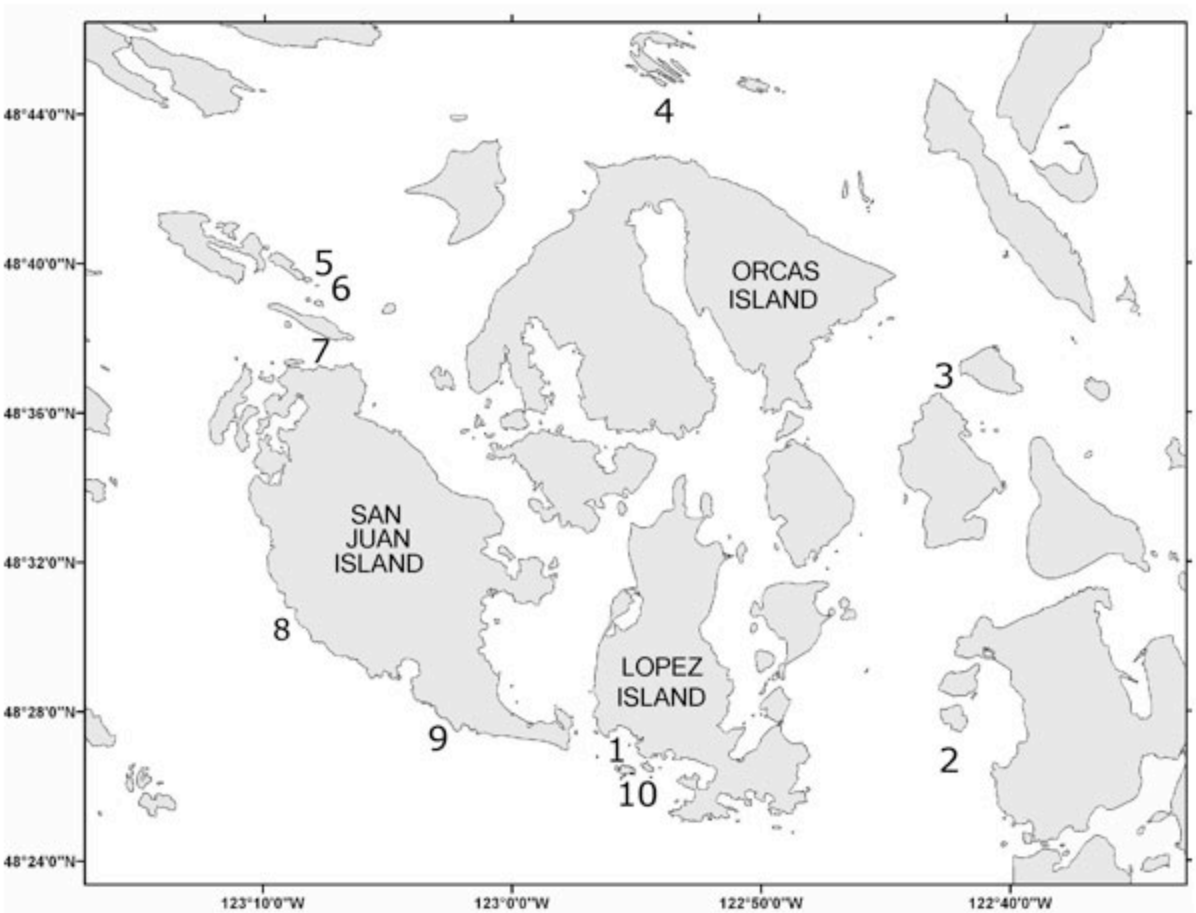


Figure 1. Map of WDFW *Haliotis kamtschatkana* index stations established in 1992 in the San Juan Archipelago, Washington State (Figure produced by WDFW and used with permission, methods according to Rothaus et al. 2008).

Trends

The decline of pinto abalone in Washington State has been of concern since the early 1990s (Rothaus et al. 2008). While commercial harvest of abalone has never been permitted in the state, the sport fishery may have extracted as many as 38, 200 individuals per year in the San Juan Archipelago (Bargmann 1984). It is therefore possible that abalone densities may have already been too low for successful fertilization or recruitment at the time of the sport fishery closure in 1994. WDFW listed the pinto abalone as a candidate species for protection in 1998 and NOAA Fisheries listed it as a federal species of concern in 2004. In 2008, WDFW identified pinto abalone as a Species of Greatest Conservation Need. In British Columbia, Canada, pinto abalone were uplisted to endangered in 2009, where populations are generally found at higher densities than Washington stocks (COSEWIC 2009).

The WDFW index site surveys in the San Juan Archipelago were repeated in 1994, 1996, 2003, 2004, 2005, 2006 and 2009. These surveys indicate a decline in abalone abundance of 83% from

1992 to 2009 (WDFW)(methods according to Rothaus et al. 2008)(Figure 2). Rothaus et al. (2008) also found an increase in mean shell length of 10.4 mm between 1992 and 2006, indicating a substantial shift in the size distribution of abalone populations, a pattern also present in the most recent survey in 2009(WDFW)(methods according to Rothaus et al. 2008)(Figure 3). This signifies a shift in abalone population age structure from younger to older animals, indicative of repeated recruitment failure (Rothaus et al. 2008). Recruitment failure following substantial declines in abalone density have been demonstrated elsewhere, for example in British Columbia, Canada (Tomascik and Holmes 2003) and in California (e.g., Miner et al. 2006). In Washington, the observed increases in mean shell length oppose the notion that the observed populations declines are a result of continued illegal harvest, because poaching is likely to result in a shift in length frequency toward smaller individuals (Rothaus et al. 2008). Pinto abalone populations may be unlikely to recover without intervention (Rothaus et al. 2008). Since 2004, a program of hatchery-based rearing and outplanting aimed at restoring abalone populations in Washington State has been led by the Puget Sound Restoration Fund (PSRF) and several local partners. In the summer of 2009, nearly 2,000 abalone were outplanted near Anacortes and Port Angeles, Washington.

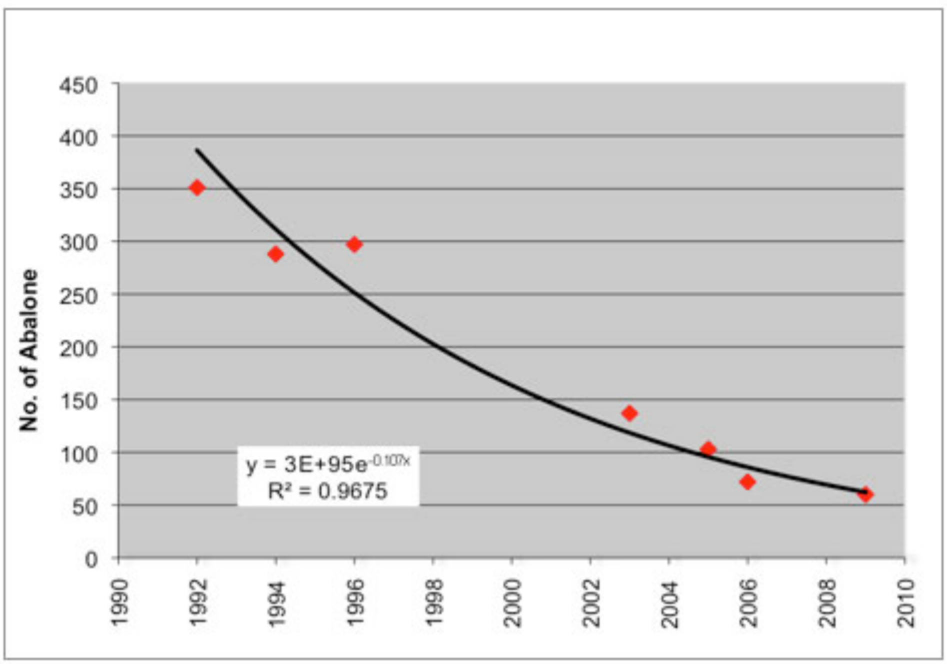


Figure 2. Pinto abalone abundance in the San Juan archipelago. Trends in abundance at 10 index stations from 1992 to 2009 (Figure produced by WDFW from unpublished data used with permission; methods according to Rothaus et al. 2008).

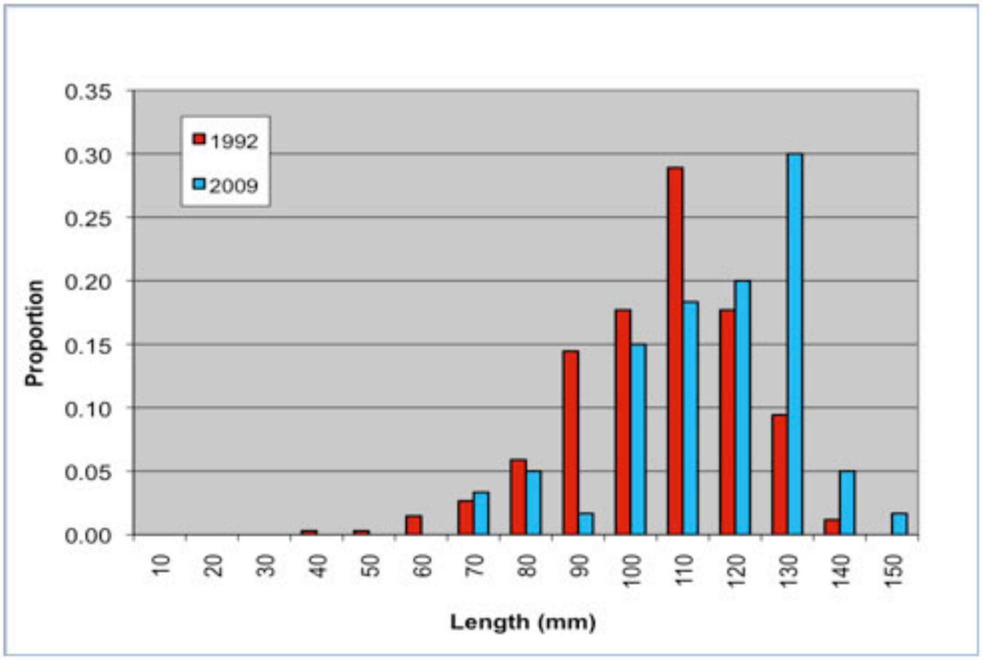


Figure 3. Pinto abalone shell length frequency in the San Juan archipelago. Trends in shell length from 10 index sites from 1992 to 2009 (Figure produced by WDFW from unpublished data used with permission; methods according to Rothaus et al. 2008).

Uncertainties

Many aspects of abalone biology and ecology are not well understood yet may be important in explaining both the decline and the recovery potential for pinto abalone in the Puget Sound region. While recreational fisheries likely played a role in the decline of pinto abalone in the San Juan Islands, the relative importance of harvesting and other factors is not known. While predation, habitat preferences, food availability and abiotic conditions will all likely affect the success of restoration efforts, the extent to which each of these factors may limit abalone populations is not well understood.

Summary

Pinto abalone are in severe decline in Puget Sound waters and are presently at densities where they may not be self-sustaining. Monitoring at index stations in the San Juan Islands showed an 83% decrease in abundance since 1992 despite their listing as federal species of concern, state candidate species, and the cessation of recreational harvest in 1994. Shell length surveys reveal that the population of pinto abalone in the San Juan Islands is aging without replacement although the direct causes of this recruitment failure warrant continued investigation. The long-term success of current hatchery-based rearing and outplanting programs is unknown at this time as efforts were recently initiated over the last five years.

Literature Cited

Babcock, R., and J. Keesing. 1999. Fertilization biology of the abalone *Haliotis laevigata*: laboratory and field studies. Canadian Journal of Fisheries and Aquatic Sciences 56:1668-1678.

Bargmann, G. G. 1984. Recreational diving in the State of Washington and the associated harvest of food fish and shellfish. State of Washington, Dept. of Fisheries, Marine Fish Program, Olympia, WA.

Bouma, J. V. 2007. Early life history dynamics of pinto abalone (*Haliotis kamtschatkana*) and implications for recovery in the San Juan archipelago, Washington State. Thesis (M S). School of Aquatic and Fisheries Science, University of Washington, Seattle, WA.

Campell, A. E. 2000. Workshop on rebuilding abalone stocks in British Columbia. Pages 173-185 in Can. Spec. Publ. Fish. Aquat. Sci. .

COSEWIC. 2009. COSEWIC assessment and update status report on the Northern Abalone *Haliotis kamtschatkana* in Canada. Committee on the Status of Endangered Wildlife in Canada, Ottawa, CAN.

Miner, C. M., J. M. Altstatt, P. T. Raimondi, and T. E. Minchinton. 2006. Recruitment failure and shifts in community structure following mass mortality limit recovery prospects of black abalone. Marine Ecology-Progress Series 327:107-117.

Rothaus, D. P., B. Vadopalas, and C. S. Friedman. 2008. Precipitous declines in pinto abalone (*Haliotis kamtschatkana kamtschatkana*) abundance in the San Juan Archipelago, Washington, USA, despite statewide fishery closure. Canadian Journal of Fisheries and Aquatic Sciences 65:2703(2709).

Dungeness Crabs

Background

Dungeness crabs (*Cancer magister*) occur throughout Washington waters, including the outer coast (mostly in coastal estuaries) and inland waters. Dungeness crabs use different habitats throughout their life cycle: as larvae they are planktonic, as juveniles they are found in intertidal mixed sand or gravel areas with algae or eelgrass (Holsman et al. 2006) and as adults they are found in subtidal or intertidal areas on sand, mud, or associated with eelgrass beds. Bare habitats are infrequently used by juveniles, most likely due to a lack of refuge from predation and decreased food abundance (McMillan et al. 1995). Vegetated, intertidal estuaries appear to be important nursery habitats for young crabs (Stevens and Armstrong 1984); older crabs have been shown to move progressively into unvegetated subtidal channels (Dinnel et al. 1986, Dethier 2006).

Annual settlement and survival of Dungeness crabs are typically variable. This variation stems from biotic factors such as predation and food availability, as well as abiotic factors such as water temperature and currents that transport larvae away from or toward nearshore areas. However, recruitment variability of Puget Sound populations is less than that seen in coastal populations (McMillan et al. 1995, Dethier 2006). There is evidence for local retention of Dungeness crab larvae within Puget Sound with a smaller proportion of recruits originating from coastal or oceanic stocks although this ratio is likely to vary from year to year (Dinnel et al. 1993, McMillan et al. 1995). Furthermore, the degree to which larvae originating in Puget Sound are transported through oceanic water before re-entering the sound is not well understood (Dethier 2006).

As predators and scavengers, Dungeness crabs feed upon a broad range of prey including small mollusks, crustaceans, clams, and fishes. They also prey for a wide variety of taxa, which varies with their life history stage. Larvae are preyed upon by coho and Chinook salmon and rockfishes; juveniles by a wide variety of fishes; and adults by fishes, seals, octopuses, and each other (generally when molting) (Orcutt et al. 1976, Reilly 1983, Dethier 2006).

Threats to Dungeness crabs include: low dissolved oxygen, variation in temperature and salinity, fisheries, habitat alteration or loss, and pollutants such as insecticides, hydrocarbons from oil spills and heavy metals. Because juvenile crabs rely on estuarine habitats and are also potentially more sensitive to toxins, early life history stages are likely to be more influenced by human activities (Dethier 2006).

StaTus

Due to their dependence on estuaries as juveniles, their value as recreational, commercial and tribal resources and their vulnerability to a suite of human impacts, Dungeness crab are included in the Washington Department of Fish and Wildlife (WDFW) Priority Habitats and Species List (Fisher and Velasquez 2008). However, there is currently no monitoring of Dungeness crab populations in Puget Sound that enable a reconstruction of population trends, status and sustainable harvest rates. Instead, time series of landings are used to gauge trends in population size over time. Commercial harvest quotas and recreational harvest season duration are determined from pre-season surveys that assess the relative abundance of mature females. The fishery is a male-only fishery, with a 6.25" (15.875 cm) carapace width minimum size. It is difficult to know whether temporally stable harvest rates represent stable population sizes or reflect changes in harvest effort or regulations (de Mutsert et al. 2008) Indeed, the increases in recreational landings may reflect increased fishing effort from a growing human population.

The current recommendations for Dungeness crab management in Puget Sound by WDFW include the reduction of habitat degradation by development, reduction in pollutants, and the reduction of impacts of fisheries (Fisher and Velasquez 2008)

Trends

Landings of Dungeness crab in Puget Sound have been highly variable, peaking at more than two million pounds in the late 1970s, declining in the 1980s, and rising again from the 1990s to 2005 (Dethier 2006). From 1995 to 2005, the biomass of Dungeness crab harvested annually by commercial, recreational, and tribal groups has shown an increase from six million pounds per season to approximately eight million pounds per season (Figure 1)(WDFW catch data, reported in Dethier 2006, PSP 2007) Increases in landings can reflect either an increase in fishing pressure or an increase in the abundance of the resource.

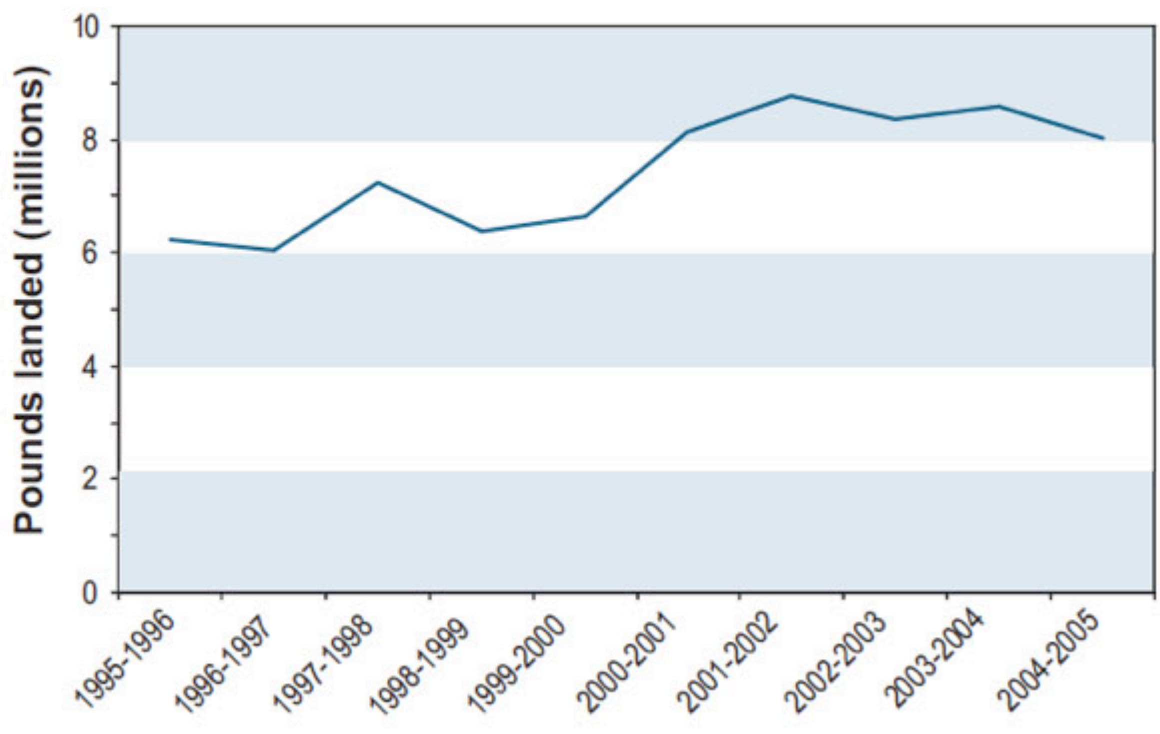


Figure 1. Dungeness crab harvest (commercial, recreational and tribal) landings from 1995 to 2005. (WDFW catch data, reported in Dethier 2006, PSP 2007) http://wdfw.wa.gov/fish/shelfish/crab/historic.htm).

Uncertainties

Because fisheries landings can be influenced by variables such as fishing effort that do not necessarily reflect crab population abundances, trends in landings data are not considered a reliable indicator of population status (de Mutsert et al. 2008). WDFW has estimated Dungeness crab abundance using a closed ring pot survey from 1999 to the present, however data from this survey have not been published.

Summary

Like many marine species with complex life histories, Dungeness crabs occupy different ecological niches throughout their life cycle and in therefore rely on multiple intact habitats. The associations between crabs and estuarine habitats, particularly nearshore habitats for juveniles may link habitat abundance and condition to the long-term health of Puget Sound Dungeness crabs. While landings data provide some information about the status of the fishery, they are not a reliable way to estimate natural population levels or trends.

Literature Cited

de Mutsert, K., J. H. Cowan, T. E. Essington, and R. Hilborn. 2008. Reanalyses of Gulf of Mexico fisheries data: Landings can be misleading in assessments of fisheries and fisheries ecosystems. Proceedings of the National Academy of Sciences 105:2740-2744.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Puget Sound Nearshore Partnership Report No. 2006-04, Seattle, WA.

Dinnel, P. A., D. A. Armstrong, and R. O. McMillan. 1993. Evidence for multiple recruitment-cohorts of Puget Sound Dungeness crab, Cancer magister. Marine Biology 115:53-63.

Dinnel, P. A., O. Russell, D. A. McMillan, D. A. Armstrong, T. C. Wainwright, A. J. Whiley, R. Burge, and R. Bumgarner. 1986. Padilla Bay Dungeness crab, *Cancer magister*, habitat study. Washington Department of Ecology, Seattle, WA.

Fisher, W., and D. Velasquez. 2008. Management recommendations for Washington's priority habitats and species: Dungeness crab (*Cancer magister*). Olympia, WA.

Holsman, K. K., P. S. McDonald, and D. A. Armstrong. 2006. Intertidal migration and habitat use by subadult Dungeness crab *Cancer magister* in a NE Pacific estuary. Marine Ecology-Progress Series 308:183-195.

McMillan, R. O., D. A. Armstrong, and P. A. Dinnel. 1995. Comparison of intertidal habitat use and growth-rates of 2 northern Puget Sound cohorts of 0+ age Dungeness crab, *Cancer magister*. Estuaries 18:390-398.

Orcutt, H. G., R. N. Tasto, P. W. Wild, C. W. Haugen, and E. E. Ebert. 1976. Dungeness crab research program. California Department of Fish and Game.

PSP. 2007. 2007 Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Olympia, WA.

Reilly, P. N. 1983. Predation on Dungeness crabs, *Cancer magister*, in Central California. California Department of Fish and Game, Sacramento, CA.

Stevens, B. G., and D. A. Armstrong. 1984. Distribution, abundance, and growth of juvenile dungeness crabs, *Cancer magister*, in Grays Harbor estuary, Washington. U.S National Marine Fisheries Service Fisheries Bulletin 82:469-483.

Jellyfish
Background

The term jellyfish is taxonomically broad, referring to gelatinous plankton in the phyla Ctenophora (comb jellies) and Cnidaria (all other jellyfish). While jellyfish have been components of pristine marine ecosystems for millennia, recent worldwide increases in the abundance of some jellyfish have been associated with anthropogenic perturbations such as eutrophication (Arai 2001), overfishing (Lynam et al. 2006), climate warming (Mills 2001, Lynam et al. 2004, Purcell 2005), and coastal development (Richardson et al. 2009). Because many jellyfish have a complex life history that includes free-living sexual and asexual phases, populations can increase rapidly when environmental conditions change to favor them.

Jellyfish blooms can disrupt human activities such as fishing, recreational beach use, and power plant operations (Purcell et al. 2007, Richardson et al. 2009). Moreover, jellyfish blooms can substantially alter food webs (e.g., Ruzicka et al. 2007, Pauly et al. 2009) by decreasing energy flow to higher trophic levels (Richardson et al. 2009) and by altering community composition of lower trophic levels through selective feeding (Purcell et al. 2007). Notably, the high degree of diet overlap between jellyfish and forage fish such as herring (Purcell and Arai 2001, Brodeur et al. 2008) is thought to be a driver of observed increases in jellyfish abundances in systems where forage fish are removed (Lynam et al. 2006). After such removals, fish recovery can be impeded by jellyfish predation on eggs and juvenile phases of their fish competitors (Purcell and Arai 2001), effectively preventing the reestablishment of fish populations (Lynam et al. 2006). Chum salmon (*Oncorhynchus keta*) are one of the few reported predators of jellyfish that occur in Puget Sound (Purcell and Arai 2001, Rice 2007)

Status

Data pertaining to jellyfish abundance in Puget Sound are scarce, but information is growing (Rice 2007, Reum et al. 2010). Biomass estimates determined from surface-towed trawl surveys conducted at 52 sites in Puget Sound in 2003 revealed relative abundances of jellyfish as high as 80% to 90% of the total trawl biomass at multiple sites in both the South Sound and in the Main Basin (Rice 2007)(Figure 1). By contrast, the observed relative abundances in the more northern regions of the Whidbey Basin and Rosario Strait were generally much lower (Figure 1). Importantly, when basin-wide data were considered, Rice (2007) noted an apparent inverse relationship between fish and jellyfish biomass. The jellyfish species observed were the Scyphomedusae *Cyanea capillata*, *Phacellophora camschatica*, *Aurelia* sp., the Hydromedusa *Aequorea* sp., and the Ctenophore *Pleurobrachia bachei* (Rice 2007). In June and September of 2007, Reum et al. (2010) conducted a more taxonomically-detailed study using bottom trawls in the northern and southern portions of Hood Canal (Hazel Point and Hoodsport, respectively) and in the Whidbey Basin (Useless Bay and Possession Sound). The species they reported were *Phacellophora camtschatica*, *Cyanea capillata*, *Aurelia labiata* and *Aequorea victoria*. While the abundances of jellyfish were both temporally and spatially variable, Reum et al. (2010) found that abundances were generally highest in June and at the southern portion of the Hood Canal mainstem near Hoodsport (Figure 2).

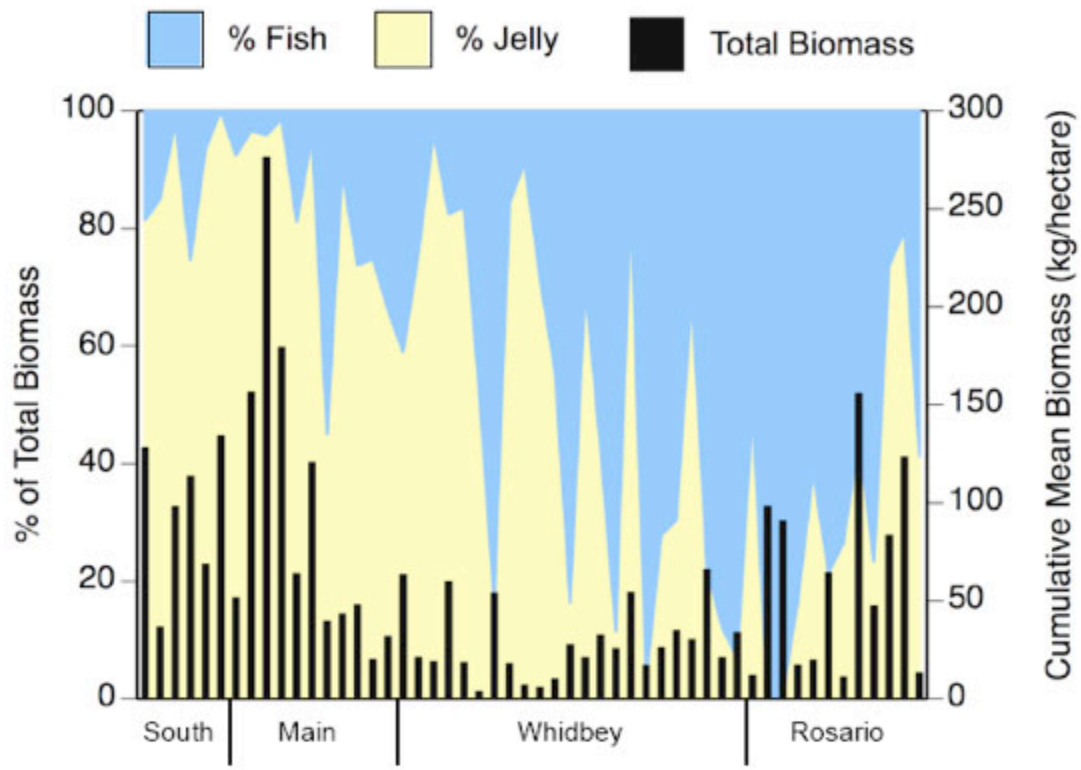


Figure 1. Percentage fish (blue area) and jelly (yellow area) in the total biomass (black bars) for sites within each region. Each bar is the sum of the four monthly means from May to August for each site. Reprinted with permission from Rice (2007).

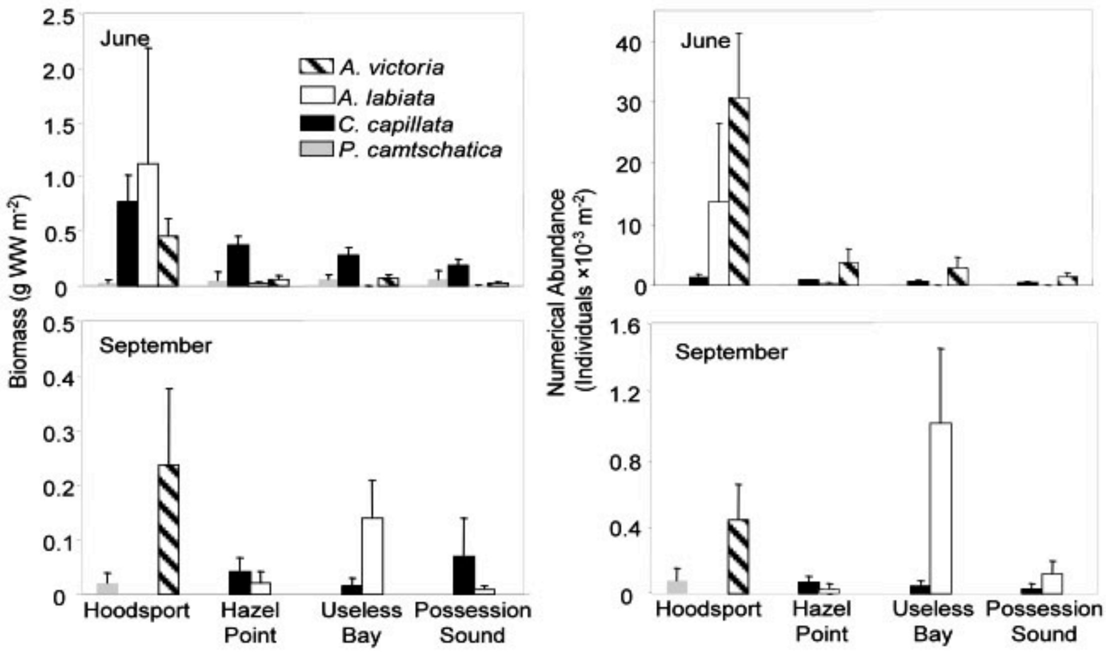


Figure 2. Biomass and numerical abundance densities sampled in June and September at four locations in Puget Sound, WA. Note that the y-axis for biomass and numerical abundances are scaled differently between June and September to better visualize variation in species composition. Error bars indicate standard deviation. Reprinted with permission from Northwest Science (Reum et al. 2010).

Trends

At this time it is not possible to determine temporal trends in jellyfish abundance in Puget Sound because existing data were collected using different methods and at different locations.

Uncertainties

The biology and ecology of most jellyfish are poorly known. In particular, knowledge of the asexually reproducing benthic polyp phase is limited (Boero et al. 2008). While it is clear from the limited available data that jellyfish are present in Puget Sound and that the likely causes of jellyfish outbursts (e.g., eutrophication, climate warming, coastal development and fishing pressure) also occur in Puget Sound to varying degrees, whether these factors are leading to increased jellyfish abundances has not been investigated. Because jellyfish have few predators, there is a high potential for them to disrupt food webs by displacing forage fish and other mid-trophic consumers, which could cause dramatic changes to the Puget Sound ecosystem. Indeed, a recent analysis of food webs in other temperate marine systems conducted by Samhouri et al. (2009) found that jellyfish were strongly correlated with multiple important ecosystem attributes, particularly those pertaining to trophic energy transfer.

Summary

While the direct mechanisms responsible for increases in jellyfish abundance in other marine systems are still being elucidated (Mills 2001, Purcell et al. 2007, Boero et al. 2008, Richardson et al. 2009), there appear to be associations between anthropogenically-perturbed systems and increased jellyfish abundance. The existing data are not sufficient to assess temporal patterns of jellyfish abundance in Puget Sound. Improved monitoring of spatial and temporal variability in jellyfish abundance as well as variation likely abiotic drivers would help to elucidate the causes and potential consequences of changing jellyfish abundance.

Literature Cited

Arai, M. N. 2001. Pelagic coelenterates and eutrophication: a review. Hydrobiologia 451:69-87.

Boero, F., J. Bouillon, C. Gravili, M. P. Miglietta, T. Parsons, and S. Piraino. 2008. Gelatinous plankton: irregularities rule the world (sometimes). Marine Ecology Progress Series 356:299-310.

Brodeur, R. D., C. L. Suchman, D. C. Reese, T. W. Miller, and E. A. Daly. 2008. Spatial overlap and trophic interactions between pelagic fish and large jellyfish in the northern California Current. Marine Biology (Berlin) 154:649-659.

Lynam, C. P., M. J. Gibbons, B. E. Axelsen, C. A. J. Sparks, J. Coetzee, B. G. Heywood, and A. S. Brierley. 2006. Jellyfish overtake fish in a heavily fished ecosystem. Current Biology 16:R492-R493.

Lynam, C. P., S. J. Hay, and A. S. Brierley. 2004. Interannual variability in abundance of North Sea jellyfish and links to the North Atlantic Oscillation. Limnology and Oceanography 49:637-643.

Mills, C. E. 2001. Jellyfish blooms: Are populations increasing globally in response to changing ocean conditions? Hydrobiologia:55-68.

Pauly, D., W. Graham, S. Libralato, L. Morissette, and M. Deng Palomares. 2009. Jellyfish in ecosystems, online databases, and ecosystem models. Hydrobiologia 616:67-85.

Purcell, J. E. 2005. Climate effects on formation of jellyfish and ctenophore blooms: a review. Journal of the Marine Biological Association of the UK 85:461-476.

Purcell, J. E., and M. N. Arai. 2001. Interactions of pelagic cnidarians and ctenophores with fish: a review. Hydrobiologia 451:27-44.

Purcell, J. E., S.-i. Uye, and W.-T. Lo. 2007. Anthropogenic causes of jellyfish blooms and their direct consequences for humans: a review. Marine Ecology Progress Series 350:153-174.

Reum, J., M. Hunsicker, and C. Paulsen. 2010. Species composition and relative abundance of large medusae in Puget Sound, WA. Northwest Science.84(2):131-140. doi: 10.3955/046.084.0202 .

Rice, C. A. 2007. Evaluating the biological condition of Puget Sound. Ph.D. dissertation. University of Washington, Aquatic and Fishery Sciences, Seattle, WA.

Richardson, A. J., A. Bakun, G. C. Hays, and M. J. Gibbons. 2009. The jellyfish joyride: causes, consequences and management responses to a more gelatinous future. Trends in Ecology & Evolution 24:312-322.

Ruzicka, J. J., R. D. Brodeur, and T. C. Wainwright. 2007. Seasonal food web models for the oregon inner-shelf ecosystem: Investigating the role of large jellyfish. California Cooperative Oceanic Fisheries Investigations Reports 48:106-128.

Samhouri, J., P. Levin, and C. Harvey. 2009. Quantitative evaluation of marine ecosystem indicator performance using food web models. Ecosystems 12:1283-1298.

Forage Fishes

Background

Forage fishes are small schooling fishes that form a critical link in the marine food web between zooplankton and larger fish and wildlife consumers. They occupy every marine and estuarine nearshore habitat in Washington, and much of the intertidal and shallow subtidal areas of the Puget Sound Basin are used by these species for spawning habitat. Status of forage fish populations can be an indicator of the health and productivity of nearshore systems (PSP 2009). Information on forage fish life history, distribution, and habitat preferences is summarized in Marine Forage Fishes of Puget Sound (Penttila 2007) and the Forage Fish Management Plan (Bargmann 1998).

The three most common forage fish species in the Puget Sound basin are Pacific herring (*Clupea pallasi*), surf smelt (*Hypomesus pretiosus*), and Pacific sand lance (*Ammodytes hexapterus*), and are therefore the focus of this section.

Pacific Herring

Pacific herring are a pelagic fish species found from northern Baja California to northern Honshu Island, Japan. They are found throughout the Puget Sound basin and are a mix of "resident" and "migratory" stocks (Gao et al. 2001, Penttila 2007, Stick and Lindquist 2009). Migratory populations cycle between the winter spawning grounds in the inside waters and the mouth of the Strait of Juan de Fuca in the summer, while resident stocks reside in the inside waters year-round (Penttila 2007). The faster individual growth rates observed in some herring populations are thought to be the result of fish leaving Puget Sound to feed in more productive oceanic waters and thus help to differentiate between migratory and resident stocks. For example, the Squaxin Pass herring population has a slower growth rate and is classified as "resident" while the Cherry Point population has a faster growth rate and is classified as "migratory" (Stick and Lindquist 2009).

Herring spawning occurs between January and April, with the majority of spawning taking place in February and March. Herring become ready to spawn over a two-month period by moving from deep water into shallow nearshore areas. The large natural and decadal oscillations in herring stock abundance are reflected in the area of spawning used annually. Most spawning areas appear to have "outlier" areas, used only during periods of high stock abundance, and "core" areas, used during periods of low stock abundance (Penttila 2007). Herring spawn on benthic marine macro-vegetation such as eelgrass or red macroalgae in the shallow subtidal and low intertidal region. Herring spawn preferentially in sheltered bays as opposed to vegetation beds on adjacent open shorelines (Stick and Lindquist 2009)(Figure 1).

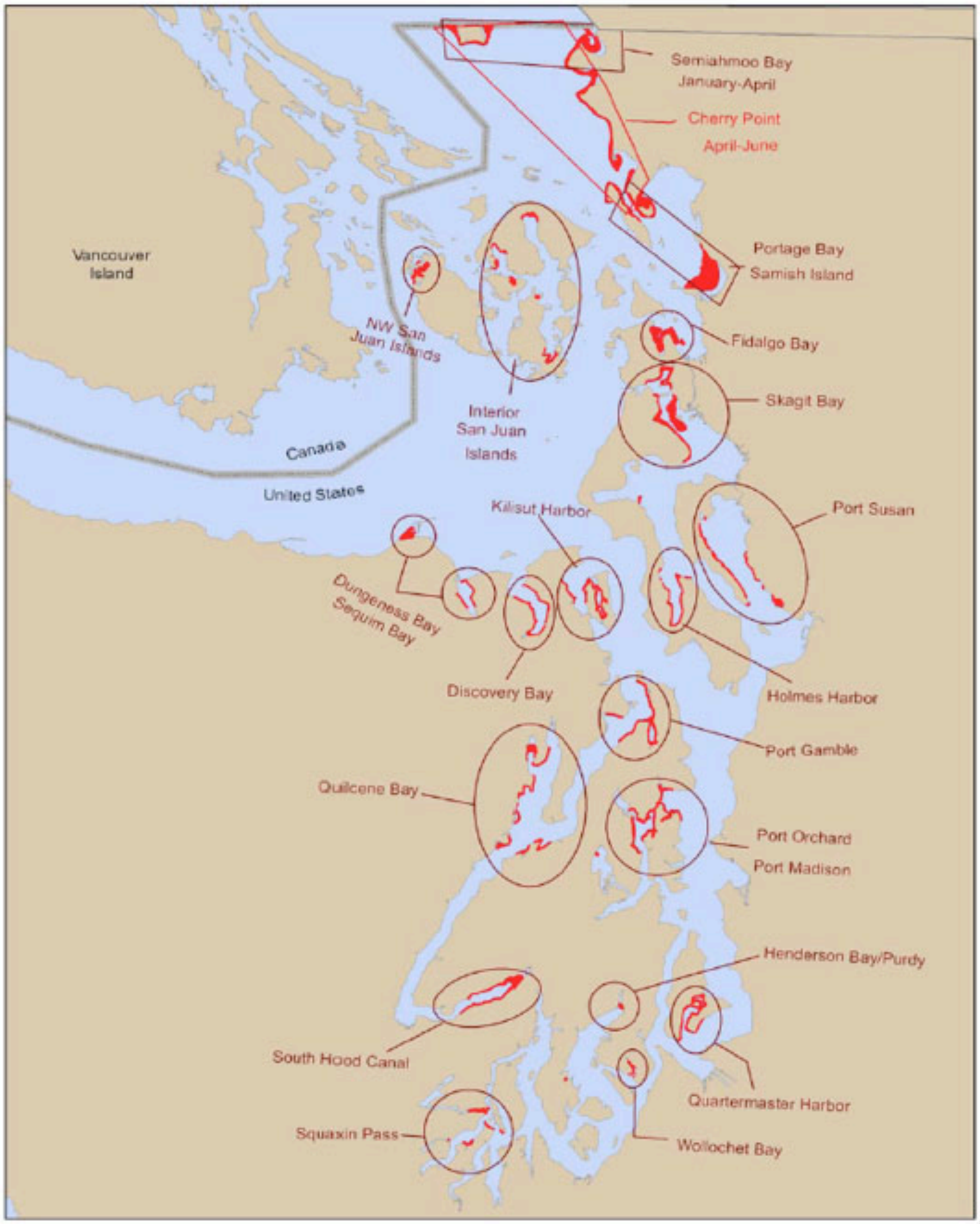


Figure 1. Documented Pacific herring spawning areas in Puget Sound (reprinted from Stick and Lindquist 2009 with permission from Washington Department of Fish and Wildlife).

Within the Puget Sound basin, autonomous stocks of herring are defined as having geographically distinct spawning areas and seasons. Two herring populations are deemed genetically distinct from other Puget Sound: the Cherry Point population which is distinctive for its late spawn timing (Small et al. 2005, Beacham et al. 2008, PSP 2008) and the Squaxin Pass population (Stick and Lindquist 2009)(Figure 1), which is thought to be spatially isolated from other populations (Small et al. 2005). Other sampled herring stocks show no evidence of genetic distinction (Small et al. 2005, Beacham et al. 2008), suggesting that these stocks may be part of a metapopulation where sufficient gene flow reduces genetic divergence (Stick and Lindquist 2009). If Puget Sound herring stocks act as a metapopulation, it may be more relevant to examine abundance trends on a larger scale than individual stock level, with Cherry Point and Squaxin Pass being the exceptions (Stick and Lindquist 2009).

Surf Smelt

Surf smelt are a nearshore species found from Long Beach, California to Chignik Lagoon, Alaska. They occur throughout the marine waters of Washington and in the southernmost region of Puget Sound. For the duration of their lifespan, surf smelt appear to inhabit shallow nearshore zones in the general area of their spawning (Penttila 2007).

Surf smelt spawning habitat is distributed throughout the Puget Sound basin and over a broad variety of conditions (e.g., variable salinity or shading). Spawning areas are usually occupied during summer (May-August), fall-winter (September-March), or year-round (monthly spawning with a seasonal peak)(Bargmann 1998, Penttila 2007). Spawning beaches are used on an annual basis, and as with Pacific herring, surf smelt have been shown to utilize "outlier" spawning sites during periods of high stock abundance (Penttila 2007).

Surf smelt use predictable shoreline areas for spawning across seasons; all spawning beaches first mapped by the Washington Department of Fish and Wildlife (WDFW) in the 1930s are still used by surf smelt. The critical habitat elements for spawning are substrate and tidal elevation. Surf smelt spawn in the uppermost one-third of the tidal range and most beaches appear suitable for surf smelt spawning habitat ranging from sheltered beaches to fully-exposed pebble beaches (Penttila 2007). Due to the diffuse nature of surf smelt spawning habitat there are no obvious grounds for stock definition in geographical terms.

Pacific Sand Lance

The Pacific sand lance occurs throughout the coastal northern Pacific Ocean from the Sea of Japan to southern California, and is widespread within the nearshore marine waters of Washington, including the entire Puget Sound basin. Sand lances inhabit nearshore waters and spawn between November and February. Sites and spawning habitats of sand lance are similar to that of surf smelt: upper intertidal sand and gravel beaches. Sand lance spawning often takes place on beaches at the distal ends of drift-cells, where accretionary shoreforms tend to occur. Because sand lance and surf smelt deposit eggs in the upper intertidal, they are particularly vulnerable to shoreline habitat modifications (Bargmann 1998).

Status

Of the forage fishes reviewed in this document, only Pacific herring populations have been monitored with sufficient detail to permit status evaluation. Surf smelt and sand lance populations are generally not considered threatened or endangered yet their abundances are currently unknown (Penttila 2007, PSP 2007).

Because of the dependence of forage fish on specific macro-vegetation for spawning, both environmental conditions and human activity (e.g., nearshore development) are likely to affect forage fish spawning biomass (Penttila 2007, Stick and Lindquist 2009). For this and other reasons (e.g., the difficulty in sampling adult populations), regulations have focused on managing forage fish spawning habitat. The Washington Administrative Code (WAC) (220-110), state Growth Management Act (GMA), and WDFW Priority Habitats and Species Program (PHS) all identify forage fish habitat as priority conservation "critical areas" or "areas of concern" for forage fish management (Penttila 2007).

Pacific Herring

There are 19 different stocks of Pacific herring in Puget Sound, based on timing and location of spawning activity (Bargmann 1998, PSP 2007). For 2007-2008, less than half of Puget Sound herring stocks were classified as healthy or moderately healthy (Stick and Lindquist 2009)(Table 1). This is similar to the status breakdown for the previous two-year periods (2003-04, 2005-06). The combined spawning biomass for all Puget Sound, excluding Cherry Point, is considered moderately healthy compared to the previous 25-year mean (11,656 tons for 2007-08 compared with 16,263 tons for 25-year mean). The abundance of south and central Puget Sound herring stocks, excluding Squaxin Pass (which is considered healthy at this time), are considered moderately healthy for 2007-08 (Stick and Lindquist 2009)(Table 1). The cumulative north Puget Sound regional spawning biomasses are considered depressed. Cherry Point continues to be considered critical; spawning biomass decreased during 2007 and 2008. Fidalgo Bay has also declined significantly since 1999 (Stick and Lindquist 2009)(Table 1). The Strait of Juan de Fuca regional status has generally been classified as critical, primarily due to Discovery Bay and Dungeness/Sequim Bay stocks suffering serious declines in biomass in recent years (Table 1) (Penttila 2007, PSP 2007, Stick and Lindquist 2009).

Table 1. Puget Sound herring stock status based on previous 2-year mean abundance compared to previous 25-year mean abundance (from Stick and Lindquist 2009).

STOCK STATUS - Describes a stock's current condition based primarily on recent (previous 2 year mean) abundance compared to long-term (previous 25 year mean) abundance.
Stock criteria such as survival, recruitment, age composition, and spawning ground habitat condition are also considered.
HEALTHY - A stock with recent two year mean abundance above or within 10% of the 25 year mean.
MODERATELY HEALTHY - A stock with recent two year mean abundance within 30% of the 25 year mean, and/or with high dependence on recruitment.
DEPRESSED - A stock with recent abundance well below the long term mean, but not so low that permanent damage to the stock is likely (i.e., recruitment failure).
CRITICAL - A stock with recent abundance so low that permanent damage to the stock is likely or has already occurred (i.e., recruitment failure).
DISAPPEARANCE - A stock which can no longer be found in a formerly consistently utilized spawning ground.
UNKNOWN - Insufficient assessment data to identify stock status with confidence.

Region / Stock	2008	2006	2004	2002	2000	1998	1996	1994
South Central Puget Sound	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY
Squaxin Pass	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY
Wollochet Bay	UNKNOWN	UNKNOWN	UNKNOWN	UNKNOWN				
Quartermaster Harbor	DEPRESSED	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY
Port Orchard-Madison	HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	DEPRESSED	DEPRESSED	DEPRESSED
South Hood Canal	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	UNKNOWN	UNKNOWN
Quilcene Bay	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	UNKNOWN
Port Gamble	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	DEPRESSED	HEALTHY	HEALTHY
Kilisut Harbor	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	UNKNOWN	HEALTHY
Port Susan	MOD. HEALTHY	DEPRESSED	DEPRESSED	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	DEPRESSED	MOD. HEALTHY
Holmes Harbor	HEALTHY	HEALTHY	HEALTHY	HEALTHY	DEPRESSED	HEALTHY	UNKNOWN	UNKNOWN
Skagit Bay	HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY	UNKNOWN
North Puget Sound	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY
Fidalgo Bay	DEPRESSED	DEPRESSED	DEPRESSED	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY
Samish/Portage Bay	HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	HEALTHY	MOD. HEALTHY
Interior San Juan Is.	DEPRESSED	MOD. HEALTHY	DEPRESSED	MOD. HEALTHY	DEPRESSED	UNKNOWN	UNKNOWN	UNKNOWN
N.W. San Juan Is.	DISAPPEARANCE	DEPRESSED	CRITICAL	DEPRESSED	UNKNOWN	DEPRESSED	UNKNOWN	UNKNOWN
Semiahmoo Bay	MOD. HEALTHY	MOD. HEALTHY	DEPRESSED	MOD. HEALTHY	DEPRESSED	DEPRESSED	HEALTHY	HEALTHY
Cherry Point	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	DEPRESSED	MOD. HEALTHY
Strait of Juan de Fuca	CRITICAL	DEPRESSED	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL
Discovery Bay	CRITICAL	DEPRESSED	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL	CRITICAL
Dungeness/Sequim Bay	DEPRESSED	DEPRESSED	DEPRESSED	MOD. HEALTHY	HEALTHY	HEALTHY	HEALTHY	UNKNOWN
Puget Sound Combined	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	HEALTHY	MOD. HEALTHY	MOD. HEALTHY	MOD. HEALTHY	HEALTHY
Individual Stock Comparison	**2008**	**2006**	**2004**	**2002**	**2000**	**1998**	**1996**	**1994**
HEALTHY	6 stocks	6 stocks	4 stocks	8 stocks	10 stocks	7 stocks	7 stocks	4 stocks
MOD. HEALTHY	3 stocks	4 stocks	5 stocks	7 stocks	2 stocks	3 stocks	2 stocks	5 stocks
DEPRESSED	6 stocks	7 stocks	6 stocks	1 stock	3 stocks	5 stocks	3 stocks	1 stock
CRITICAL	2 stocks	1 stock	3 stocks	2 stocks	2 stocks	2 stocks	1 stock	1 stock
DISAPPEARANCE	1 stock	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks	0 stocks
UNKNOWN	1 stock	1 stock	1 stock	1 stock	1 stock	1 stock	5 stocks	7 stocks
	47%	56%	50%	83%	71%	59%	69%	82%
	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy	Healthy or Mod. Healthy

Trends

Pacific Herring

The cumulative spawning biomass of all Puget Sound herring stocks, except the Cherry Point stock, has fluctuated between about 10,000 to 16,000 tons (PSP 2009, Stick and Lindquist 2009) (Figure 2). Stocks in south and central Puget Sound have exhibited a general increasing trend, however this may be due to increased sampling effort since 1996. If the abundance of stocks are assumed to be at their mean levels during years when data are not available, then the estimated aggregate population sizes in the south and central Puget Sound stocks are comparable to those from 1970s and 1980s. Stocks in northern Puget Sound, excluding the Cherry Point stock, have remained at a low level of abundance (PSP 2009, Stick and Lindquist 2009) (Figure 2). Similarly, herring spawning biomass in the Strait of Juan de Fuca region continues to be very low and with the exception of 2006, the Discovery Bay herring stock has decreased steadily to between 200-250 tons annually since the mid 1990s (Stick and Lindquist 2009).

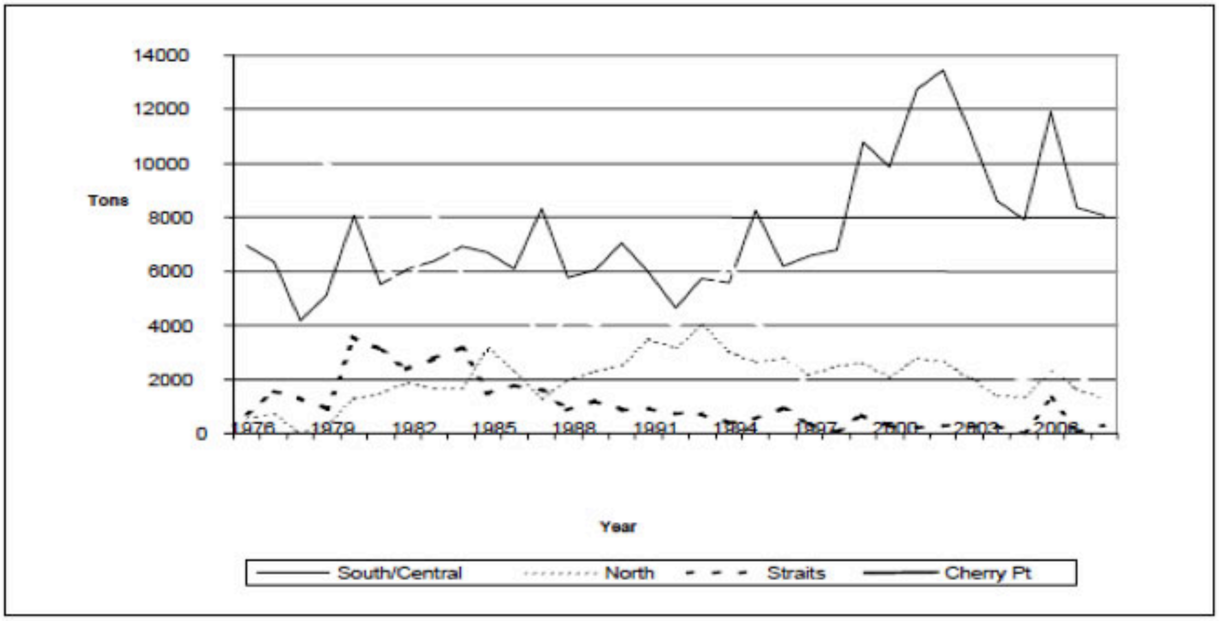


Figure 2. Estimated Puget Sound herring total spawning biomass by region and Cherry Point stock, 1976-2008 (data from Stick and Lindquist 2009, reprinted from PSP 2009).

Puget Sound herring stock abundance is significantly affected by mortality rates, which can be attributed to fishing and natural mortality (Stick and Lindquist 2009) (Figure 3). The mean estimated annual natural mortality rate for sampled Puget Sound herring stocks (excluding Cherry Point) since 1990 has averaged 72%, compared with typical mortality rates of 30-40% for herring worldwide. The Cherry Point herring stock annual mortality rate has increased to an average of 68% since 1990. Fishing mortality has averaged about 4% of estimated natural mortality since 1997. Predation, disease, and climatic changes are all potential causes of increased natural mortality (Stick and Lindquist 2009).

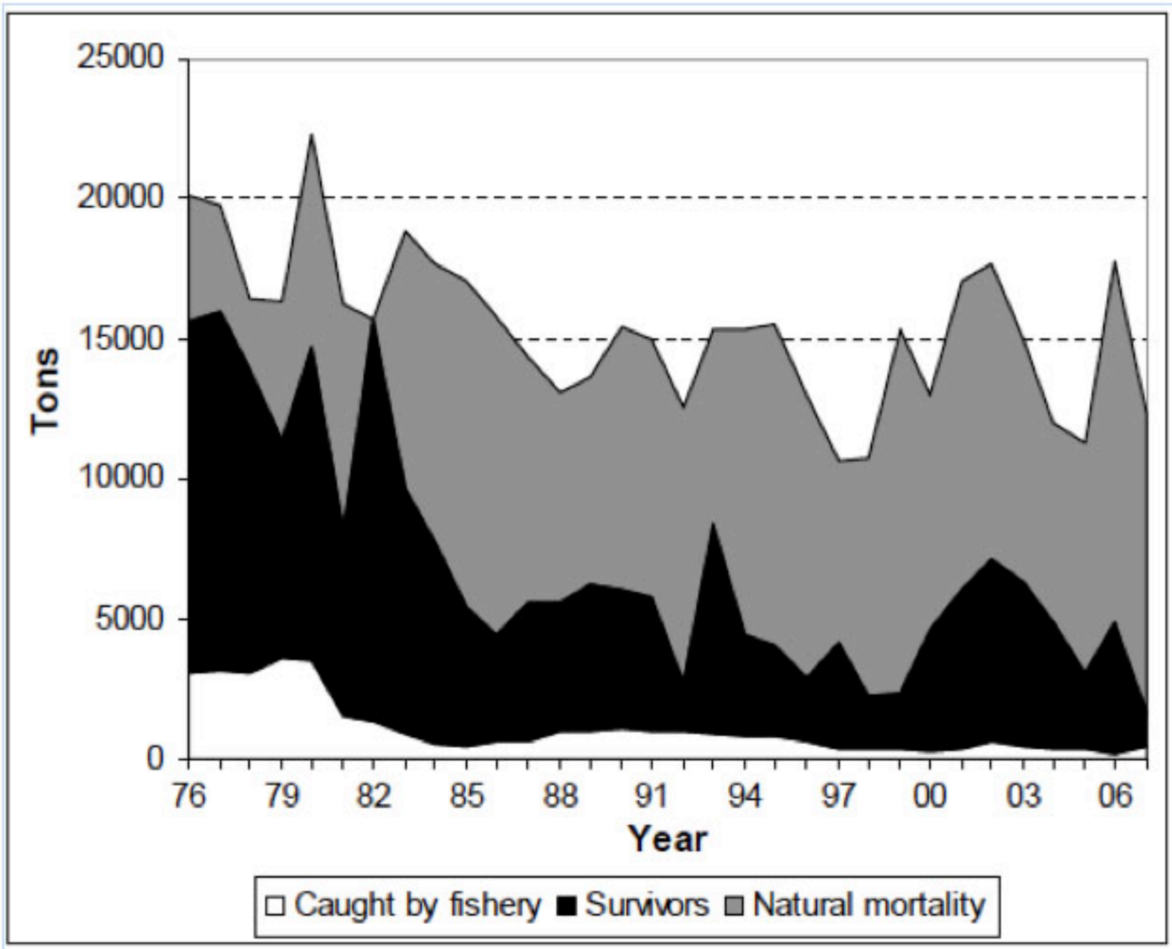


Figure 3. Annual tonnage estimates of herring in Puget Sound determined by natural mortality/survival rates, fishery harvest, and cumulative spawning biomass from 1976-2007 (reprinted from Stick and Lindquist 2009 with permission from Washington Department of Fish and Wildlife).

Uncertainties

Since the amount of data collected and the methods used for data collection differ across herring stocks and from year to year, Stick and Lindquist (2009) developed a system to evaluate the quality of the available information for each stock. They designated stocks which had a continuous time series of both acoustic-trawl and spawn deposition data as having “Good” data quality, stocks which had a continuous time series of only spawn deposition data as having “Fair” data quality, and populations for which there was an incomplete time series for either type of data as having “Poor” data quality. The majority of stocks assessed in this manner fell into the “Fair” category, with the best and most consistent data coming from Port Orchard/Madison and Cherry Point (Stick and Lindquist 2009)(Table 2).

Table 2. Puget Sound herring stock data quality determined by the amount of stock assessment data (evaluated in Stick and Lindquist 2009).

South/Central Puget Sound	Data Quality
Squaxin Pass	Fair
Wollochet Bay	Poor
Quartermaster Harbor	Fair
Port Orchard/Madison	Good
South Hood Canal	Poor
Quilcene Bay	Fair/Poor
Port Gamble	Fair
Kilisut Harbor	Fair/Poor
Port Susan	Fair
Holmes Harbor	Fair
Skagit Bay	Fair
North Puget Sound	
Fidalgo Bay	Fair
Samish/Portage Bay	Poor
Interior San Juan Islands	Poor
Northwest San Juan Island	Poor
Semiahmoo Bay	Fair
Cherry Point	Good
Strait of Juan de Fuca	
Discovery Bay	Fair
Dungeness/Sequim Bay	Poor

Good: A continuous time series of acoustic-trawl data & spawn deposition data.

Fair: A continuous time series of spawn deposition data only.

Poor: An incomplete time series of either type of stock assessment data.

Summary

Because of their reliance on near-shore habitats, the continued viability of these populations depends on the preservation of this habitat. Pacific Herring have a complicated population structure based on differences in the location and timing of spawning, although only two stocks are deemed genetically distinct. Data on population status are most extensive for Pacific Herring stocks, where current status and trends are mixed. The previously large Cherry Point stock is severely depressed from historical population levels. The prospect that this stock is now regulated by diseases has been raised and remains an active area of research. Long term assessment of other major species is needed to evaluate their current population levels and trends so that the impacts of habitat loss, fishing and climate change can be determined.

Literature Cited

Bargmann, G. 1998. Forage Fish Management Plan. Washington Department of Fish and Wildlife, Olympia, WA.

Beacham, T. D., J. F. Schweigert, C. MacConnachie, K. D. Le, and L. Flostrand. 2008. Use of microsatellites to determine population structure and migration of Pacific Herring in British Columbia and adjacent regions. Transactions of the American Fisheries Society 137:1795-1811.

Gao, Y. W., S. H. Joner, and G. G. Bargmann. 2001. Stable isotopic composition of otoliths in identification of spawning stocks of Pacific herring (*Clupea pallasi*) in Puget Sound. Canadian Journal of Fisheries and Aquatic Sciences 58:2113-2120.

Penttila, D. 2007. Marine Forage Fishes in Puget Sound. Puget Sound Nearshore Partnership Report No. 2007-03, Seattle, WA.

PSP. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Page 260, Olympia, WA.

PSP. 2008. Species and Biodiversity Topic Forum. Puget Sound Partnership, Seattle, WA.

PSP. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Small, M. P., J. L. Loxterman, A. E. Frye, J. F. Von Bargen, C. Bowman, and S. F. Young. 2005. Temporal and spatial genetic structure among some Pacific Herring populations in Puget Sound and the Southern Strait of Georgia. Transactions of the American Fisheries Society 134:1329-1341.

Stick, K. C., and A. Lindquist. 2009. 2008 Washington State Herring Stock Status Report. Stock Status Report No. FPA 09-05, Olympia, WA.

Bentho-Pelagic Fish

Background

Bentho-pelagic fish utilize both demersal (bottom) habitats and shallower portion of the water column, often as part of diel migrations whereby fish feed in shallow water at night and move to deeper water to form schools during the day. Four currently or historically important species of bentho-pelagic fish in Puget Sound are the Pacific hake (*Merluccius productus*), the Pacific cod (*Gadus macrocephalus*), the Walleye pollock (*Theragra chalcogramma*) and the spiny dogfish (*Squalus acanthias*). Three of these species (Pacific hake, Pacific cod and Walleye pollock) were included in a petition for federal listing under the Endangered Species Act in 1999.

Pacific hake

Pacific hake (also known as Pacific whiting) form three spawning stocks in the Northeast Pacific: a coastal, highly migratory stock, a Strait of Georgia stock and a Puget Sound stock. Currently the two inland stocks and the coastal stock are federally recognized as Distinct Population Segments (DPS) based on genetic, demographic and behavioral differences (Gustafson et al. 2000), however more recent genetic evidence suggests further subdivision between southern Puget Sound and Strait of Georgia populations may be warranted (Iwamoto et al. 2004). In Puget Sound, Pacific hake form large seasonal spawning aggregations in Port Susan which was the target of a substantial fishery for many years (Pedersen 1985). Spawning activity has been also reported in Dabob Bay (Bailey and Yen 1983). Spawning in Puget Sound is thought to occur primarily from February to April (Gustafson et al. 2000). Pacific hake produce pelagic eggs which develop into larvae that feed primarily on copepods (McFarlane and Beamish 1985). As juvenile and small adults, the diet of hake is primarily euphausiid crustaceans which also undergo diel migrations (e.g., Mackas et al. 1997). Large adults consume a wide array of prey including amphipods, squid, Pacific herring, crabs, shrimp and juvenile Pacific hake (McFarlane and Beamish 1985, Gustafson et al. 2000). Pacific hake are also important prey for a suite of predators; this group includes walleye Pollock, Pacific cod, rockfish, spiny dogfish and marine mammals such as sea lions (McFarlane and Beamish 1985, Gustafson et al. 2000). Pacific hake in Puget Sound are believed to reach maturity at approximately 30 cm and 4-5 years of age; they can live for up to 20 years and reach sizes of 73 cm. The size at maturity and average body size of Pacific hake Puget Sound are reported to have decreased in Pacific hake from the 1980s to 2000 (WDFW data)(reported in Gustafson et al. 2000).

Pacific cod

Pacific cod occur in the Northeast Pacific occur from Alaska to California. Adult cod typically occupy deep habitats (50 – 300 m) and have been historically been observed forming spawning aggregations at multiple locations throughout Puget Sound (Palsson 1990, Gustafson et al. 2000). Although the review conducted by Gustafson et al. (2000) did not find conclusive evidence of population differentiation of North American Pacific cod, more recent otolith (Gao et al. 2005) and microsatellite (Cunningham et al. 2009) studies suggest that inland (Strait of Georgia and Puget Sound) populations are distinct from the coastal stocks. Pacific cod typically mature at 2-3 years of age at approximately 45 cm, with an estimated maximum lifespan of 18 years. Pacific cod occupy different habitats throughout their life cycle. Eggs are typically found in demersal habitats followed by a transition to the pelagic zone as larvae and small juveniles, settling to intertidal or subtidal sand or eelgrass habitats as large juveniles and moving to deep water as adults (reviewed by Gustafson et al. 2000). Juvenile cod feed on crustaceans such as shrimp, mysids and amphipods; the diet of adults is though to reflect the relative availability of prey with some preference for walleye pollock in large (>70 cm) adults (Gustafson et al. 2000). Pacific cod are preyed upon by a variety of predators including pelagic fishes, sea birds, whales, halibut, shark and other Pacific Cod.

Walleye pollock

Walleye pollock have a similar distribution to Pacific cod (from Alaska to California) and Puget Sound is thought to be one of the southernmost spawning locations across this range although this has been not well characterized (Gustafson et al. 2000). The degree of population structure of Pacific walleye pollock remains under investigation; earlier work using microsatellites did not find evidence of genetic structure (O'Reilly et al. 2004) whereas more recent work using non-neutral alleles has found evidence for differentiation between Puget Sound and other populations across its geographic range (Canino et al. 2005). Adult pollock are typically found between waters of 100 and 300 m depth and spawn at similar depths, with a lifespan of up to 17 years and a maximum size of up to 100 cm. Pollock eggs and larvae are pelagic, while juveniles and adults feed in surface waters at night and form schools in deeper water during the day although the presence of predators has been shown to shift this behavior to an association with structure such as seagrass (Sogard and Olla 1993). Larvae feed on copepod nauplii (e.g., Canino et al. 1991) while juveniles primarily feed euphausiids and other crustaceans (e.g., Brodeur 1998). Adult walleye pollock opportunistically feed on fishes, copepods and amphipods; a recent study of fish diets in Puget Sound found that walleye pollock stomachs contents were primarily pelagic invertebrates and small mobile benthic feeders (Reum and Essington 2008). Predators of walleye pollock include seabirds, marine mammals and other fish including cannibalistic interactions (summarized in Gustafson et al. 2000).

Spiny dogfish

Spiny dogfish are cartilaginous fish in the subclass Elasmobranchii along with sharks, skates and rays and are one of the longest-lived and latest-maturing taxa within this group, with an age-at-maturity of approximately 36 years and a lifespan of nearly 100 years (Saunders and McFarlane 1993). This life history strategy makes them particularly susceptible to overharvesting. While not typically harvested for consumption, they were intensely fished in the Puget Sound region in the 1940s for their oils, which are rich in Vitamin A. They are known to consume a variety of fish including cod and herring as well as crustaceans such as crabs (Jensen 1965). In the Northeast Pacific, a recent tagging study revealed them to be quite migratory, with some individuals utilizing habitats across British Columbia, the Strait of Georgia and Western Vancouver Island (McFarlane and King 2003).

Status

Pacific hake

As a result of declines in abundance in the Puget Sound population between the 1980s and late 1990s, the inland DPS (Strait of Georgia and Puget Sound) of Pacific hake is currently listed as a Federal Species of Concern and a Washington State Candidate Species (Palsson et al. 1998, Gustafson et al. 2000). If the Puget Sound population becomes recognized as a single DPS, the level of protection may increase. Commercial and recreational fisheries for hake in Puget Sound were closed in 1991 (Gustafson et al. 2000). Current population levels of Pacific hake in Puget Sound are not known.

Pacific cod

Pacific cod are currently listed as a Washington State Candidate Species (Palsson et al. 1998). Concerns over declines prompted the closure of the bottom trawl fishery near Port Townsend and Protection Island in 1991 and a prohibition of recreating fishing takes (Gustafson et al. 2000). However, as with Pacific hake, current population levels of Pacific cod in Puget Sound are not well known but are presumed to be low based on research survey (bottom trawl) and trap catch rates.

Walleye pollock

Walleye pollock, like Pacific cod, are listed as Washington State Candidate species (Palsson et al. 1998), yet current population levels of walleye pollock in Puget Sound have not been assessed. Daily recreational bag limits were reduced to zero in 1997 (Gustafson et al. 2000). Published reports that assess population status of Puget Sound walleye pollock are not available.

Spiny dogfish

Estimates of spiny dogfish population status in Puget Sound have not been reported in any peer-reviewed documents

Trends

Pacific hake

Pacific hake have undergone a decline in Puget Sound since the early 1980s. WDFW annual hydro-acoustic surveys combined with species composition and length distributions gathered from midwater trawls revealed an estimated 85 % decrease in the Port Susan total spawning biomass from 1983 to 1999 (WDFW data)(reported in Gustafson et al. 2000)(Figure 1). Trends for the Dabob Bay (Hood Canal) spawning population have not been documented (Gustafson et al. 2000).

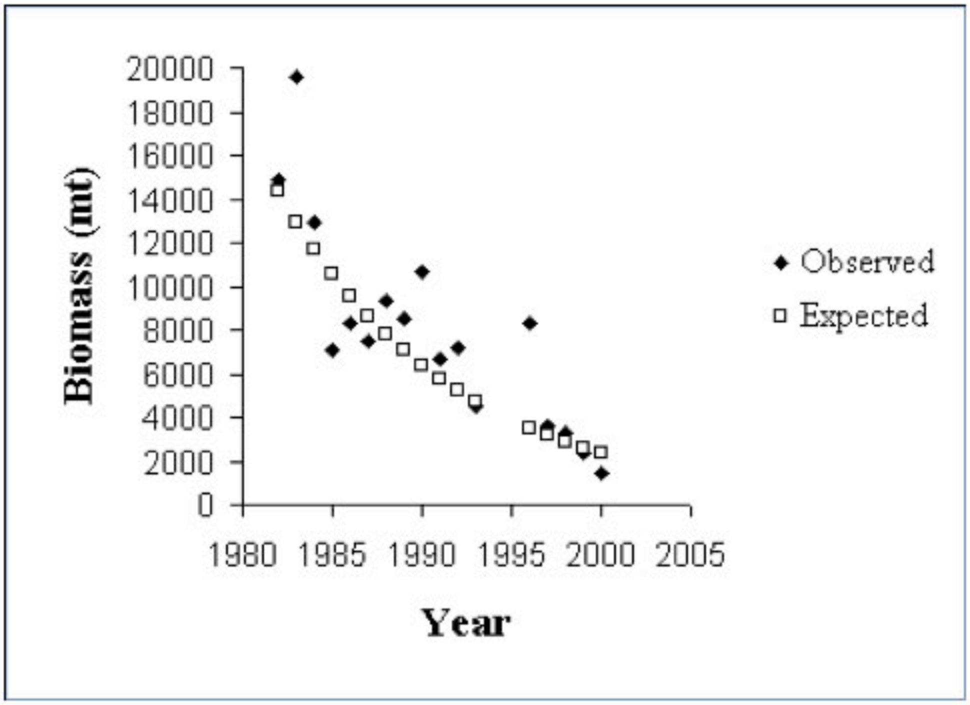


Figure 1. Results of a model predicting population declines (expected) and observed biomass of Pacific hake from WDFW trawls in Port Susan, Puget Sound from 1982 – 2000 (WDFW) (Reprinted from Gustafson et al. 2000; courtesy of NOAA Fisheries).

Pacific cod

The paucity of fishery-independent data on Pacific cod abundances makes population trends in Puget Sound difficult to assess, yet the decline in landings observed by WDFW and reported in Gustafson et al. (2000) combined with an apparent lack of subsequent reported occurrences suggest that populations in Puget Sound have likely declined substantially since the 1970s.

Walleye pollock

As with Pacific cod, the information available on walleye pollock abundance other than those based on fishery landings are lacking for Puget Sound. Fishing catches peaked in the late 1970s followed by a decline in the mid 1980s (WDFW data) (reported by Gustafson et al. 2000).

Spiny dogfish

Taylor and Gallucci (2009) report significant declines in spiny dogfish length and age at maturity and an increase in average fecundity between 1940–2000. However, these authors stressed it was difficult to discern whether these were due to density dependent effects following population declines or from climatic forcing.

Uncertainties

More information is needed to assess the current population sizes and future trends of all four key bentho-pelagic fish in Puget Sound. Specifically, analysis of long-term trends in abundance, population structure and dependence on environmental conditions is needed to ascertain status and key drivers.

Summary

Bentho-pelagic fish are important components of marine ecosystems and are often the targets of fishing pressure. In Puget Sound, Pacific hake, Pacific cod and walleye pollock were all once reported to be common and are now apparently much less abundant despite the fact that fishing pressure has been relieved. The direct causes for the declines and for the lack of rebounding are not well understood. All of these species are known to be susceptible to biophysical forcing and climatic regime shifts (Anderson and Piatt 1999, Hunt et al. 2002, Agostini et al. 2006, Agostini et al. 2008), and because Puget Sound cod and walleye pollock are at the southernmost extent of their range, these impacts may be particularly pronounced. Spiny dogfish, as an extremely long-lived shark has been shown to be susceptible to even low fishery pressure (Taylor and Gallucci 2009).

Literature Cited

Agostini, V. N., R. C. Francis, A. B. Hollowed, S. D. Pierce, C. Wilson, and A. N. Hendrix. 2006. The relationship between Pacific hake (*Merluccius productus*) distribution and poleward subsurface flow in the California Current System. Canadian Journal of Fisheries and Aquatic Sciences 63:2648-2659.

Agostini, V. N., A. N. Hendrix, A. B. Hollowed, C. D. Wilson, S. D. Pierce, and R. C. Francis. 2008. Climate-ocean variability and Pacific hake: A geostatistical modeling approach. Journal Of Marine Systems 71:237-248.

Anderson, P. J., and J. F. Piatt. 1999. Community reorganization in the Gulf of Alaska following ocean climate regime shift. Marine Ecology-Progress Series 189:117-123.

Bailey, K. M., and J. Yen. 1983. Predation by a carnivorous marine copepod, Euchaeta elongata Esterly, on eggs and larvae of the Pacific hake, *Merluccius productus*. J. Plankton Res. 5:71-82.

Brodeur, R. 1998. Prey selection by age-0 walleye pollock, *Theragra chalcogramma*, in nearshore waters of the Gulf of Alaska. Environmental Biology of Fishes 51:175-186.

Canino, M. F., K. M. Bailey, and L. S. Incze. 1991. Temporal and geographic differences in feeding and nutritional condition of walley pollock larvae *Theragra chalcogramma* in Shelikof Strait, Gulf of Alaska T. Marine Ecology-Progress Series 79:27-35.

Canino, M. F., P. T. O'Reilly, L. Hauser, and P. Bentzen. 2005. Genetic differentiation in walleye pollock (*Theragra chalcogramma*) in response to selection at the pantophysin (PanI) locus. Canadian Journal of Fisheries and Aquatic Sciences 62:2519-2529.

Cunningham, K. M., M. F. Canino, I.F.Spies, and L. Hauser. 2009. Genetic isolation by distance and localized fjord population structure in Pacific cod (*Gadus macrocephalus*): limited effective dispersal in the northeastern Pacific Ocean. Canadian Journal of Fisheries and Aquatic Sciences 66:153-166.

Gao, Y., G. Bargmann, U. Brand, and D. Noakes. 2005. Stable Isotopic and trace elemental compositions of otoliths and the stock structure of Pacific Cod, *Gadus macrocephalus*. Environmental Biology of Fishes 74:335-348.

Gustafson, R. G., W. H. Lenarz, B. B. McCain, C. C. Schmitt, W. S. Grant, T. L. Builder, and R. D. Methot. 2000. Status review of the Pacific Hake, Pacific Cod, and Walleye Polluck from Puget Sound, WA. NOAA, Department of Commerce.

Hunt, G. L., P. Stabeno, G. Walters, E. Sinclair, R. D. Brodeur, J. M. Napp, and N. A. Bond. 2002. Climate change and control of the southeastern Bering Sea pelagic ecosystem. Deep-Sea Research II 49:5821-5823.

Iwamoto, E., M. Ford, and R. Gustafson. 2004. Genetic Population Structure of Pacific Hake, *Merluccius productus*, in the Pacific Northwest. Environmental Biology of Fishes 69:187-199.

Jensen, A. L. 1965. Life history of the spiny dogfish. Fishery Bulletin 65:527-554.

Mackas, D. L., R. Kieser, M. Saunders, D. R. Yelland, R. M. Brown, and D. F. Moore. 1997. Aggregation of euphausiids and Pacific Lake (*Merluccius productus*) along the outer continental shelf off Vancouver Island. Canadian Journal of Fisheries and Aquatic Sciences 54:2080-2096.

McFarlane, G. A., and R. J. Beamish. 1985. Biology and fishery of Pacific whiting, *Merluccius productus* in the Strait of Georgia Marine Fisheries Review 47:23-34.

McFarlane, G. A., and J. R. King. 2003. Migration patterns of spiny dogfish (*Squalus acanthias*) in the North Pacific Ocean. Fishery Bulletin 101:358-367.

O'Reilly, P. T., M. F. Canino, K. M. Bailey, and P. Bentzen. 2004. Inverse relationship between F-ST and microsatellite polymorphism in the marine fish, walleye pollock (*Theragra chalcogramma*): implications for resolving weak population structure. Molecular Ecology 13:1799-1814.

Palsson, W. A. 1990. Pacific cod (*Gadus macrocephalus*) in Puget Sound and adjacent water: biology and stock assessment. Washington Department of Fish and Wildlife, Olyumpia, WA.

Palsson, W. A., T. J. Northup, and M. W. Barker. 1998. Puget Sound groundfish management plan (revised). Washington Department of Fish and Wildlife, Olyumpia, WA.

Pedersen, M. 1985. Puget Sound Pacific whiting, *Merluccius productus*, resource and industry: An overview. Marine Fisheries Review 47:35-38.

Reum, J., and T. Essington. 2008. Seasonal variation in guild structure of the Puget Sound demersal fish community. Estuaries and Coasts 31:790-801.

Saunders, M. W., and G. A. McFarlane. 1993. Age and length at maturity of the female spiny dogfish, *Squalus acanthias*, in the Strait of Georgia, British Columbia, Canada. Environmental Biology of Fishes 38:49-57.

Sogard, S., and B. Olla. 1993. The influence of predator presence on utilization of artificial seagrass habitats by juvenile walleye pollock, *Theragra chalcogramma*. Environmental Biology of Fishes 37:57-65.

Taylor, I. G., and V. F. Gallucci. 2009. Unconfounding the effects of climate and density dependence using 60 years of data on spiny dogfish (*Squalus acanthias*). Canadian Journal of Fisheries and Aquatic Sciences 66:351-366.

Rockfish

Background

Rockfish are bony fish in the Scorpaenid family, primarily in the genus *Sebastes*. Approximately 28 species of rockfish are reported from Puget Sound (Palsson et al. 2009), spanning a range of life-history types, habitats, and ecological niches. This diversity makes rockfish challenging to manage as a group and consequently, single-species management approaches have been recommended (Musick et al. 2000, Parker et al. 2000, Stout et al. 2001, Palsson et al. 2009, WDFW 2009). Rockfish in Pacific waters are among the most long-lived of teleost fishes and have low average annual reproductive success (Love et al. 2002). In combination, these characteristics make rockfish particularly susceptible to over-fishing. All of the rockfish in Puget Sound are classified as having Low or Very Low productivity according to definitions specified by the American Fisheries Society (AFS) (Musick 1999, Musick et al. 2000).

Rockfish have a biphasic life history in which pelagic larvae spend 1-2 months in the water column followed by settlement to benthic habitats that shift over ontogeny. In Puget Sound, settling rockfish are thought to associate with a variety of habitats including eelgrass, kelp, drift vegetation, and cobble fields, while many species as adults are found associated with deeper, high-relief rocky substrates (Palsson et al. 2009). While diet varies with species, developmental stage and location within the Sound, primary prey items for rockfish include Pacific herring, crabs, shrimp, surfperch, greenlings, and benthic invertebrates such as amphipods (reviewed by Palsson et al. 2009). In turn, rockfish, particularly as juveniles, are preyed upon by suite of predators including lingcod (Beaudreau and Essington 2007), salmonids and other fish (Palsson et al. 2009), while adults have been shown to be consumed by marine mammals such as harbor seals (Lance and Jeffries 2007).

Although rockfish larvae are pelagic, there is genetic evidence for limited dispersal within Puget Sound for the quillback (*S. maliger*) and copper (*S. caurinus*) rockfish (Seeb 1998) as well as for differentiation from coastal populations of brown rockfish (*S. auriculatus*) (Buonaccorsi et al. 2002). This degree of population structure is consistent with other genetic and otolith studies from coastal Pacific rockfish populations (Cope 2004, Miller et al. 2005, Burford 2009). Because of these findings, populations of each species of rockfish in the northern and southern portions of Puget Sound are recognized by WDFW to be separate stocks (Palsson et al. 2009) (Figure 1).

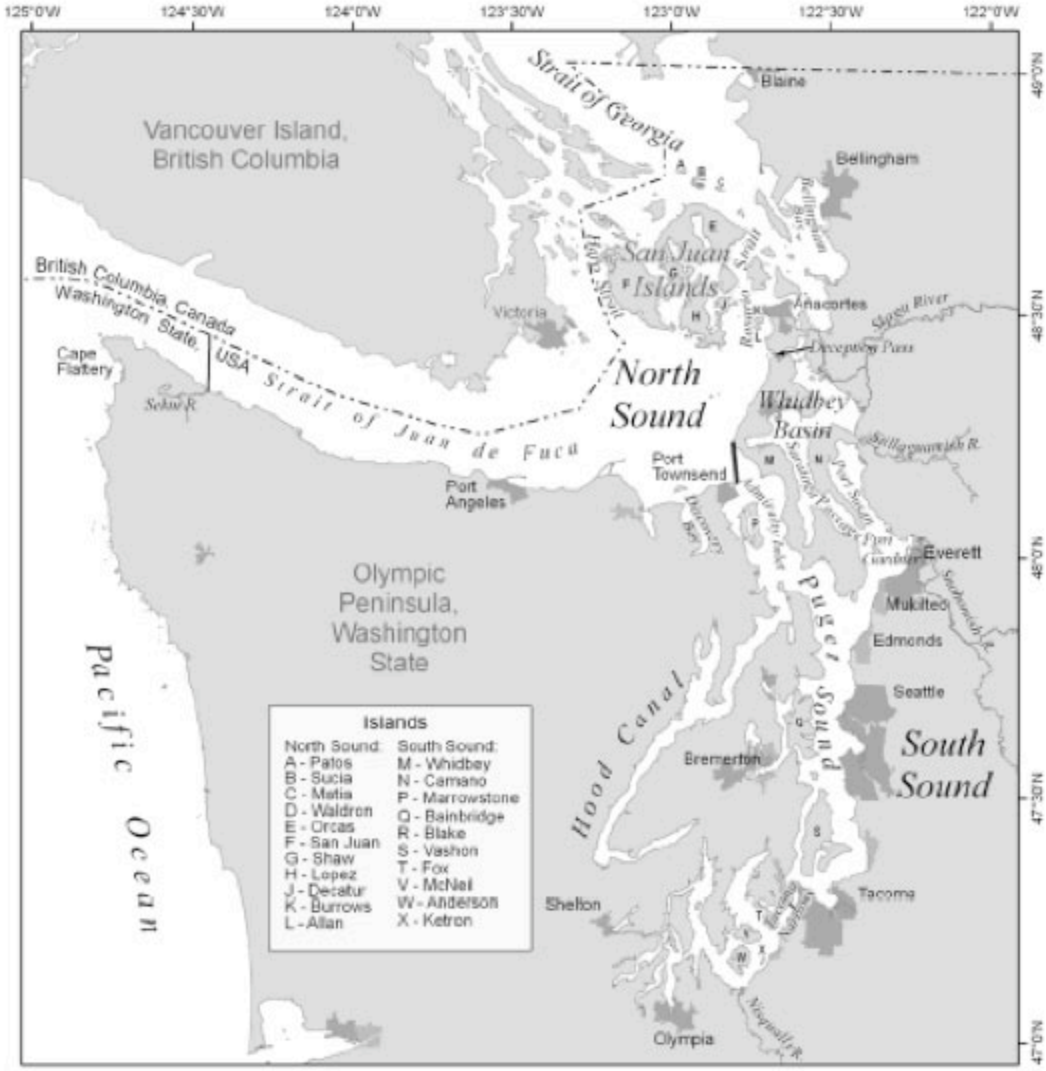


Figure 1. Map of Puget Sound showing North Sound and South Sound designations relevant to rockfish management (Reprinted from Palsson et al. 2009 with permission from .)

Currently, the Washington Department of Fish and Wildlife (WDFW) collects two types of data on rockfish in Puget Sound: those that are dependent upon information obtained from commercial and recreational fisheries (fishery-dependent data) and those that are based upon population surveys conducted by WDFW (fishery-independent data). The estimates of commercially removed biomass in Puget Sound are thought to be fairly accurate because documentation began in 1955 while recreational take has been monitored less consistently (Palsson et al. 2009). However, demographic data from the recreational fishery that inform the assessments of stock status for copper and quillback rockfish are collected by samplers trained by WDFW (Palsson et al. 2009). To obtain independent estimates of population abundances of rockfish, WDFW began conducting bottom trawls at a suite of sites in Puget Sound in 1987. The number of trawls for a given region has varied substantially over time (Palsson et al. 2009). Underwater video surveys are used to estimate biomass, density and distribution of rockfish. SCUBA transects are conducted along 30 m transects at approximately 25 sites in the North and South regions of Puget Sound (Palsson et al. 2009).

Using the abundances and trends from all available fishery-independent data, Palsson et al.(2009) classified each rockfish species as Healthy, Precautionary, Vulnerable or Depleted. These status categories are based on those used by the American Fisheries Society (Musick 1999). For the two rockfish species for which demographic data were most available (quillback and copper rockfish), designations were made based on current Spawners per Recruit (SPR) relative to 1970s SPR (proxy for an unfished population) and 1999 SPR (Palsson et al. 2009).

Status

The removal of rockfish from Puget Sound through recreational and commercial fisheries increased substantially after the Boldt Decision in 1974 when fishing restrictions were increased for salmon while rockfish were identified as a recommended alternative. Due to general declines in rockfish catches on the outer coast and to the petition for federal listing of 14 rockfish species found in the Puget Sound (Stout et al. 2001), commercial fishing for rockfish in Puget Sound has been restricted since 1999 and commercial catches have been negligible in recent years (Palsson et al. 2009). In 2002, any take of the yelloweye rockfish (*Sebastes ruberrimus*) and canary rockfish (*S. pinniger*) became prohibited. In 2004, the recreational daily limit on other rockfish species was reduced to a single fish (Palsson et al. 2009). In 2009, the Puget Sound populations of yelloweye and canary rockfish were federally listed as Threatened under the Endangered Species Act (ESA) and the bocaccio (*S. paucispinis*) was listed as Endangered. In addition to these federal listings, 14 of the 17 stocks of rockfishes in the North Puget Sound and 11 of the 15 stocks in the South Sound are currently designated by WDFW as Precautionary, Vulnerable or Depleted (Table 1).

Table 1. Summary of the status of rockfish stocks in Puget Sound (WDFW) (Palsson et al. 2009).

Species	North Sound	South Sound
Copper rockfish	Precautionary	Vulnerable
Quillback rockfish	Vulnerable	Depleted
Brown rockfish	Precautionary	Precautionary
Black rockfish	Precautionary	Precautionary
Yelloweye rockfish	Depleted	Depleted
Yellowtail rockfish	Precautionary	Precautionary
Canary rockfish	Depleted	Depleted
Bocaccio	Precautionary	Precautionary
Redstripe rockfish	Healthy	Healthy
Greenstriped rockfish	Healthy	Healthy
Splitnose rockfish	Precautionary	Precautionary
Shortspine thornyhead	Healthy	Healthy
Tiger rockfish	Precautionary	Precautionary
China rockfish	Precautionary	Not Present
Blue rockfish	Precautionary	Not Present
Vermilion rockfish	Precautionary	Precautionary
Puget Sound rockfish	Precautionary	Healthy
Number Healthy	3	4
Number Precautionary	11	7
Number Vulnerable	1	1
Number Depleted	2	3
Total Stocks Examined	17	15

A new management plan has recently been proposed by WDFW which outlines several possible management options for rockfish in Puget Sound and is currently under review (WDFW 2009). One of the key components of this plan is the recommendation that quillback, copper, black, yelloweye, bocaccio, canary and Puget Sound rockfish be managed as individual species due to their importance to recreational fisheries, conservation concerns, or ecological importance (WDFW 2009) (Table 2). In addition to these proposed changes in management, there are currently 16 marine reserves throughout Puget Sound that include the rocky habitat thought to be beneficial for rockfish.

Table 2. Proposed species of interest, habitats and reason for their selections in the Draft Puget Sound Rockfish Management Plant (WDFW).

SPECIES	COMPLEX	REASON
Copper rockfish	Nearshore	Important in recreational fisheries
Quillback rockfish	Nearshore	Important to recreational fisheries
Black rockfish	Pelagic	Important to recreational fisheries
Yelloweye rockfish	Deepwater	Conservation concerns, past economic importance
Bocaccio rockfish	Deepwater	Conservation concerns
Canary rockfish	Deepwater	Conservation concerns, past economic importance
Puget Sound rockfish (*Sebastes empheus)*	Nearshore	Important forage item

Trends

Both commercial and recreational catches of rockfishes have substantially declined since the mid 1980s and 1990s in both the North and South Puget Sound (Palsson et al. 2009) (Figure 2). Bottom trawl survey data also show declines in the harvested species of rockfishes; the two species that have increased over time (redstripe rockfish, *S. proriger* and Puget Sound rockfish, *S. emphaeus*) are smaller-bodied fish that are not harvested (Palsson et al. 2009)(Figure 3). The estimated SPR ratios for copper and quillback rockfish in the North and South Sound have also declined dramatically from 1970s to 1999 in both the North and South Sounds (Palsson et al. 2009)(Figure 4). This metric is important because it reflects the effect of fishing pressure on the reproductive capacity of a harvested population.

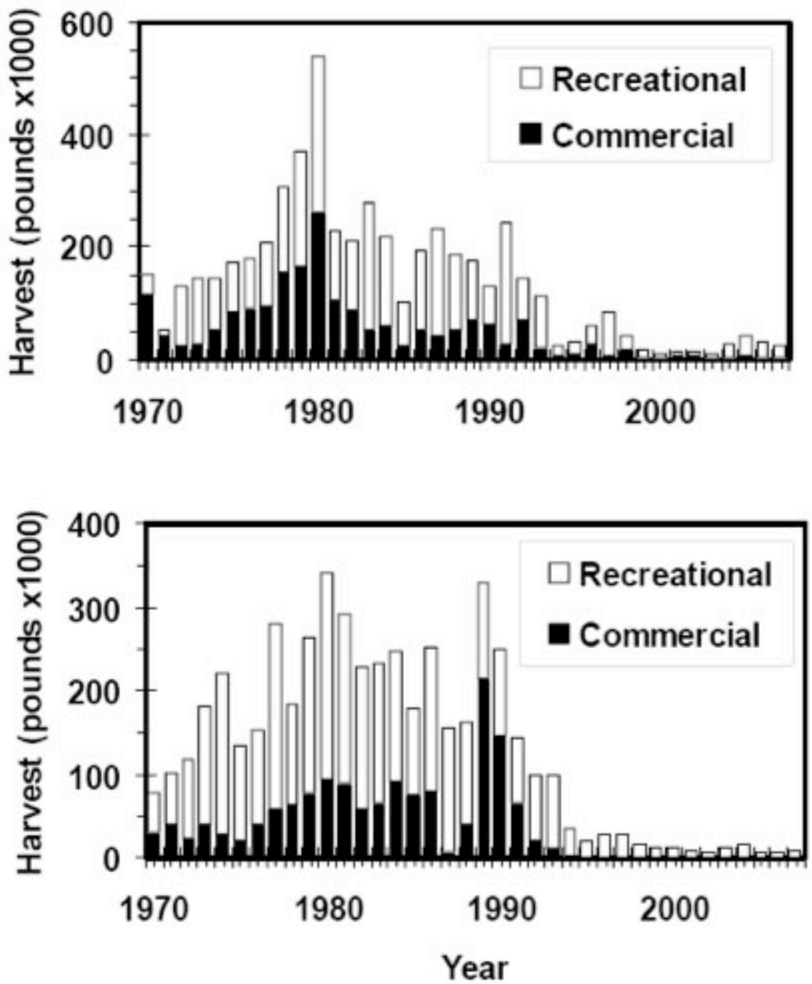


Figure 2. Total annual recreational (white) and commercial (black) harvest in pounds estimated by WDFW from North Puget Sound (top) and South Puget Sound (bottom) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)

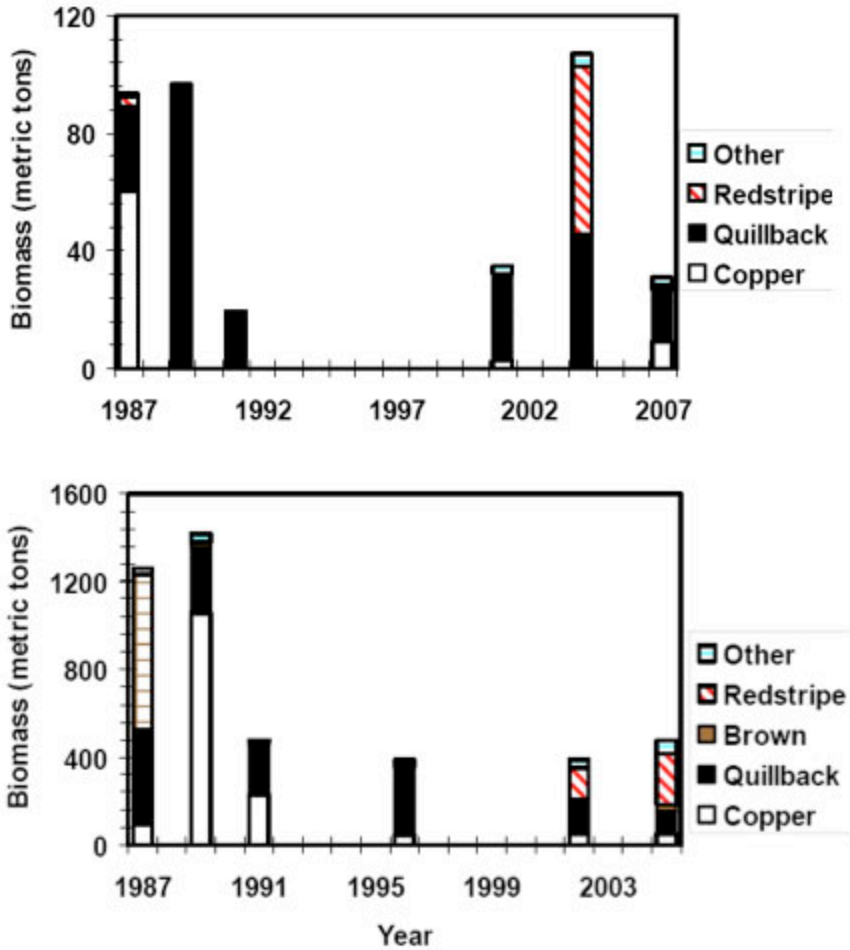
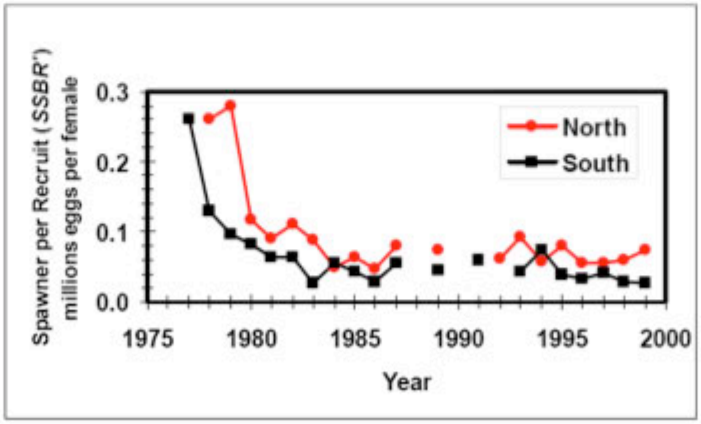


Figure 3. Biomass estimates (metric tons) from WDFW bottom trawl surveys from the Georgia Basin and East Juan de Fuca regions of North Sound (top) and South Sound (bottom) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)



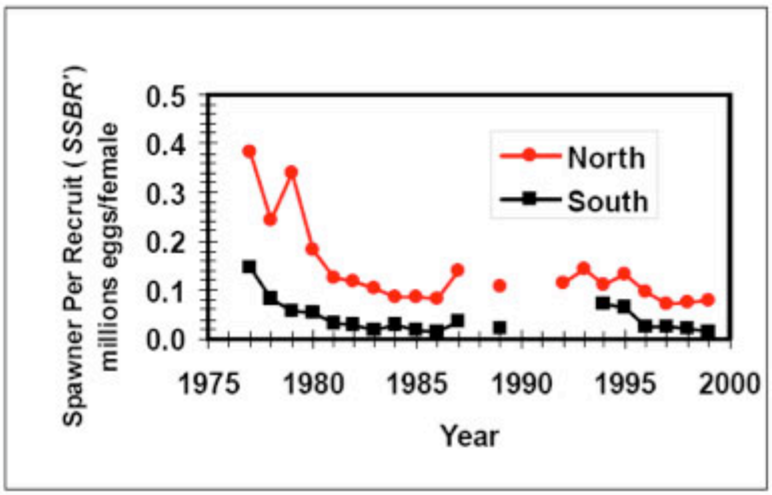


Figure 4. Spawner per Recruit Index (SSBR') for copper (top) and quillback (bottom) rockfishes in North Sound (red circles) and South Sound (black squares) (WDFW) (Reprinted from Palsson et al. 2009 with permission from Washington Department of Fish and Wildlife.)

Uncertainties

Many aspects of the ecology and biology of rockfish germane to their management in Puget Sound are not well understood. For example, ecological interactions such as predation may play important roles in determining the success of management strategies (e.g., Beaudreau and Essington 2007, Harvey et al. 2008), while demographic parameters such as age structure of populations (Berkeley et al. 2004, Berkeley 2006, Lucero 2009) or variability in the factors that drive recruitment rates are also likely to be quite important in driving the potential for rockfish recovery. Furthermore, while targeted exploitation of rockfishes in Puget Sound has diminished in recent years, the influence of continued threats such as pollution, altered food webs, incidental catch in recreational fisheries are not known.

Summary

Rockfish form a diverse assemblage of fish in Puget Sound and throughout their range. In Puget Sound, rockfish have abundances decreased substantially since quantitative monitoring began in the 1970s. These declines have resulted in the federal listing of three species under the Endangered Species Act. Because of their diversity in habitat use, ecology and life history, single-species approaches to rockfish management in Puget Sound are currently being considered.

Literature Cited

Beaudreau, A. H., and T. E. Essington. 2007. Spatial, temporal, and ontogenetic patterns of predation on rockfishes by lingcod. Transactions of the American Fisheries Society 136:1438-1452.

Berkeley, S. A. 2006. Pacific rockfish management: are we circling the wagons around the wrong paradigm? Bulletin of Marine Science 78:655-667.

Berkeley, S. A., C. Chapman, and S. M. Sogard. 2004. Maternal age as a determinant of larval growth and survival in a marine fish, *Sebastes melanops* Ecology 85:1258-1264.

Buonaccorsi, V. P., C. A. Kimbrell, E. A. Lynn, and R. D. Vetter. 2002. Population structure of copper rockfish (*Sebastes caurinus*) reflects postglacial colonization and contemporary patterns of larval dispersal. Canadian Journal of Fisheries & Aquatic Sciences 59:1375.

Burford, M. O. 2009. Demographic history, geographical distribution and reproductive isolation of distinct lineages of blue rockfish (*Sebastes mystinus*), a marine fish with a high dispersal potential. Journal of Evolutionary Biology 22:1471-1486.

Cope, J. M. 2004. Population genetics and phylogeography of the blue rockfish (*Sebastes mystinus*) from Washington to California. Canadian Journal of Fisheries & Aquatic Sciences 61:332-342.

Harvey, C. J., K. Gross, V. H. Simon, and J. Hastie. 2008. Trophic and fishery interactions between Pacific hake and rockfish: effect on rockfish population rebuilding times. Marine Ecology Progress Series 365:165-176.

Lance, M. M., and S. J. Jeffries. 2007. Temporal and spatial variability of harbor seal diet in the San Juan Island archipelago. Research Agreement No. K004431-25, Contract Report to SeaDoc Society, Olympia, WA.

Love, M. S., M. Yoklavich, and L. K. Thorsteinson. 2002. The rockfishes of the northeast Pacific. University of California Press, Berkeley, CA.

Lucero, Y. 2009. A multivariate stock-recruitment function for cohorts with sympatric subclasses: application to maternal effects in rockfish (genus *Sebastes*). Canadian Journal of Fisheries and Aquatic Sciences 66:557-564.

Miller, J. A., M. A. Banks, D. Gomez-Uchida, and A. L. Shanks. 2005. A comparison of population structure in black rockfish (*Sebastes melanops*) as determined with otolith microchemistry and microsatellite DNA. Canadian Journal of Fisheries and Aquatic Sciences 62:2189-2198.

Musick, J. A. 1999. Criteria to define extinction risk in marine fishes - The American Fisheries Society initiative. Fisheries 24:6-14.

Musick, J. A., M. M. Harbin, S. A. Berkeley, G. H. Burgess, A. M. Eklund, L. Findley, R. G. Gilmore, J. T. Golden, D. S. Ha, G. R. Huntsman, J. C. McGovern, G. R. Sedberry, S. J. Parker, S. G. Poss, E. Sala, T. W. Schmidt, H. Weeks, and S. G. Wright. 2000. Marine, Estuarine, and Diadromous Fish Stocks at Risk of Extinction in North America (Exclusive of Pacific Salmonids). Fisheries 25:6-30.

Palsson, W. A., T. S. Tsou, G. G. Bargmann, R. M. Buckley, J. E. West, M. L. Mills, Y. W. Cheng, and R. E. Pacunski. 2009. The biology and assessment of rockfishes in Puget Sound. Washington Department of Fish and Wildlife, Olympia, WA.

Parker, S. J., S. A. Berkeley, J. T. Golden, D. R. Gunderson, J. Heifetz, M. A. Hixon, R. Larson, B. M. Leaman, M. S. Love, J. A. Musick, V. M. O'Connell, S. Ralston, H. J. Weeks, and M. M. Yoklavich. 2000. Management of Pacific Rockfish. Fisheries 25:22-30.

Seeb, L. W. 1998. Gene flow and introgression within and among three species of rockfishes, *Sebastes auriculatus*, *S. caurinus*, and *S. maliger*. J Hered 89:393-403.

Stout, H. A., B. B. McCain, R. D. Vetter, T. L. Builder, W. H. Lenarz, L. L. Johnson, and R. D. Methot. 2001. Status review of Copper rockfish, Quillback rockfish and Brown rockfish in Puget Sound, Washington. NOAA Technical Memo, U.S. Department of Commerce.

WDFW. 2009. Draft Puget Sound Rockfish Conservation Plan. Washington Department of Fish and Wildlife, Olympia, WA.

Salmonids

Background

Fish in the family Salmonidae (salmon, trout, and charr) are unique in their cultural, economic and ecological role in Puget Sound. Because they utilize a very wide range of aquatic habitat types throughout their life history, they play potentially integral roles in the upland freshwater, nearshore and pelagic marine ecosystems and food webs of Puget Sound. They also provide key trophic links between habitats through their migratory behavior. While there is much variation in the behavior and ecology within and among the different salmonid species in Puget Sound, they typically use freshwater habitats to spawn, after which juveniles emerge and eventually migrate to nearshore estuaries or directly to marine pelagic habitats. Salmonids spend up to several years in marine habitats prior to returning to their natal watershed to spawn. Each life phase is thus dependent on a different suite of abiotic and biotic processes for survival. The use of nearshore habitats by juvenile salmon is thought to be a critical aspect of their capability to ultimately return and spawn (Fresh 2006). Available spawning habitat, appropriate water temperature and flow, and oceanic conditions are also important for salmonid survival and the degree of use of each type of habitat varies dramatically across the salmonid species.

The watersheds and nearshore habitats of Puget Sound currently support 8 species of salmon, trout, and charr (NOAA 2007)(Figure 1), four of which are listed as Threatened under the Endangered Species Act (ESA). These are Chinook salmon (*Oncorhynchus tshawytscha*), chum salmon (O. keta), bull trout (*Salvelinus confluentus*) and steelhead (*O. mykiss*). The recovery plan for Chinook, Hood Canal Summer Chum and bull trout put forth by Shared Strategy for Puget Sound and the Puget Sound Technical Recovery Team was adopted by NOAA Fisheries in 2007. The recovery strategy for these species is based upon the underlying principles of 1) abundance (the number of spawners); 2) productivity (the number of returning fish produced by each spawner); 3) spatial distribution (the geographic distribution of fish populations); and 4) diversity (of the genetic, physiological and morphological attributes)(NOAA 2007).

Data are collected on salmonid abundances in Puget Sound by a variety of local, state and federal agencies including Washington Department of Fish and Wildlife(WDFW), NOAA Fisheries and the US Fish and Wildlife Service. Spawner abundances are typically estimated in the field by counting the number of nests (redds) or by counting the number of spawning and/or dead fish. WDFW maintains an online database of watershed-specific spawner abundances (Salmonscape) and also conducts stock status estimates (Salmonid Stock Inventory) whereby each spawning stock is designated as Healthy, Depressed, Critical, Extinct or Unknown based upon recent abundance trends for all species except for Chinook salmon (WDFW 2002). The most recent stock inventory categorization utilized trend data from the mid 1980s to 2000 or 2001 (WDFW 2002).

Chinook salmon

The largest of the salmonids, Chinook salmon typically spawn in larger rivers and their tributaries, utilizing deeper water and larger gravel for egg burial than their congeners. While Chinook fry are often classified as either ocean-type or stream-type depending on the timing of their initial downstream migration, in Puget Sound this has further been subdivided into four broad types of strategies based upon general timing emigration from both freshwater and estuarine habitats prior to eventually migrating to coastal oceanic waters (Fresh 2006). These range from up to a year spent in natal freshwater streams with very little time spent migrating through estuarine habitat to very early emigration from freshwater with up to 120 days spent rearing in natal estuaries and tidal wetlands (summarized in Fresh 2006). This diversity is thought to be critical for the continued survival of this species (NOAA 2007). There is emerging evidence that some Chinook salmon remain in Puget Sound waters as residents with little or no time spent in oceanic waters (O'Neill and West 2009). Following entry into the open ocean via the Strait of Juan de Fuca, Chinook salmon are believed to migrate mostly northwards towards British Columbia and Alaska, remaining on the continental shelf where they typically spend 2-4 years before returning to their natal stream to spawn and die (Quinn 2005, Quinn et al. 2005).

Hood Canal Summer Chum salmon

Chum salmon typically spawn in the lower reaches of rivers with fry leaving fresh water for estuarine habitats within days of emergence. In Puget Sound, they can either remain in their natal estuaries or transition to other estuaries and nearshore habitats to rear before entering oceanic waters. While utilizing estuary habitats, chum salmon primarily feed upon epibenthic invertebrates associated with eelgrass (summarized in Fresh 2006).

Steelhead

Unlike Chinook and chum salmon, steelhead are iteroparous, displaying a diverse suite of life history variations with variable time spent in fresh, salt water and estuarine environments. They are thought to leave coastal waters immediately after entering the ocean, occupying marine habitats distinct from that of their congeners, spending 1-3 years at sea (Quinn et al. 2005, Hard et al. 2007). While little is known about the oceanic migration patterns of Puget Sound steelhead, there is evidence that they travel to the Central North Pacific (reviewed in Hard et al. 2007). The resident (non-migratory) form of steelhead (rainbow trout) is also present in Puget Sound (Hard et al. 2007).

Bull trout

Like steelhead, Bull trout are iteroparous and long lived, potentially spawning in their natal streams several times throughout their lifetime. Like cutthroat trout, bull trout often occupy nearshore marine habitats during their short seaward migration.

Status

Chinook salmon

Listed as Threatened in 1999, Chinook salmon currently maintain 22 of the estimated 30-37 historically present spawning populations that utilize rivers and streams throughout Puget Sound. (NOAA 2007)(Figure 1, Table 1). Many of the populations lost were those that spawned earlier in the spawning season when water levels are typically lower and temperatures are higher (NOAA 2007). There is also evidence that the life history variants that spend the greatest time in freshwater (stream-type) have been severely reduced in recent years, likely because these variants are particularly susceptible to watershed alteration such as damming (Beechie et al. 2006). Furthermore, the few remaining locations where the stream-type life history has persisted appear to be largely dependent on snowmelt for their water and thus may be particularly susceptible to the effects of global climate change (Beechie et al. 2006). The Puget Sound Technical Recovery Team (PSTRT) estimated the historic spawner abundances by modeling the number of individuals each watershed could support based upon habitat characteristics (NOAA 2007). For all populations for which this analysis has been conducted, current population levels are substantially less than the estimated historic levels (NOAA 2007)(Figure 2).

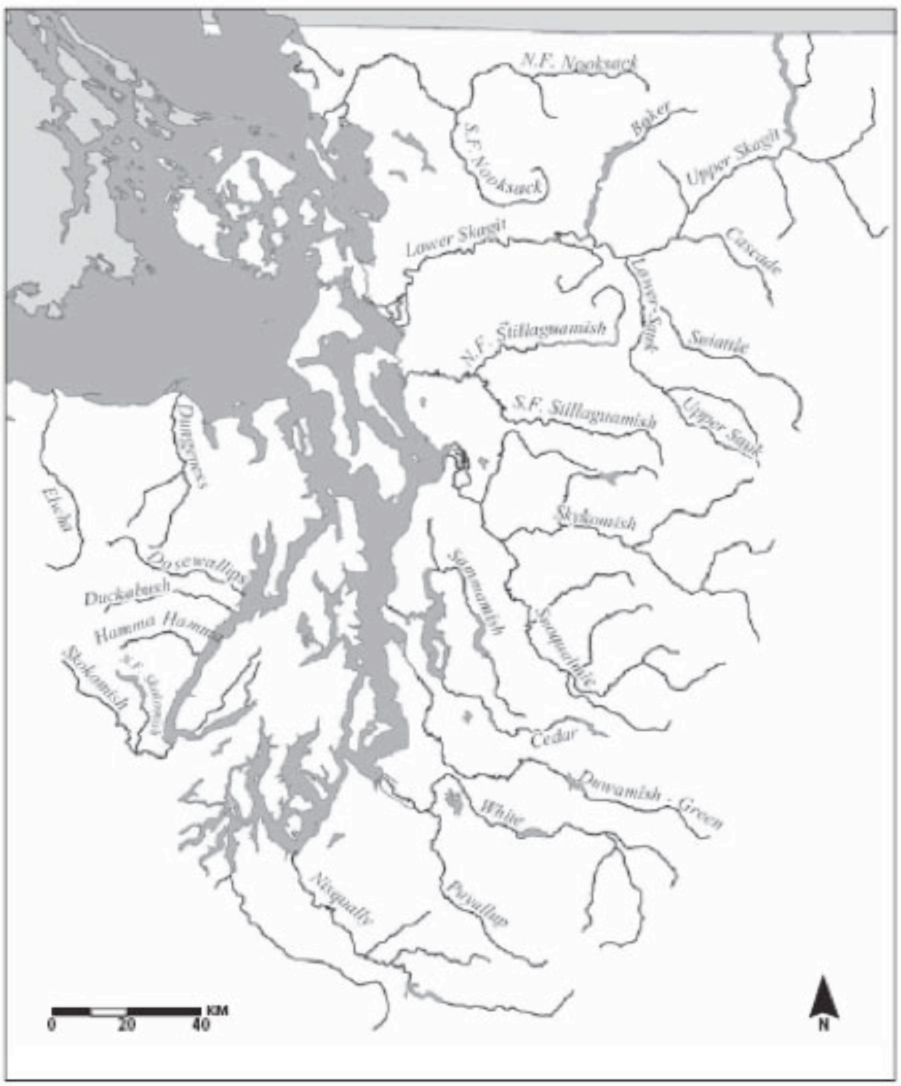


Figure 1. Map of major watersheds in Puget Sound utilized by salmonids (Reprinted from NOAA 2007; courtesy of NOAA Fisheries).

Table 1. Extant populations of Chinook salmon in Puget Sound (NOAA Salmon Recovery Plan 2007).

Geographic Region	Populations Remaining
Strait of Georgia This area includes the Nooksack River and the San Juan Islands. It is an area greatly influenced by the Fraser River and is utilized extensively for forage and migration by many Puget Sound populations.	North Fork Nooksack South Fork Nooksack
Strait of Juan de Fuca This region includes the rivers draining the north slopes of the Olympic mountains, and draining into the eastern Strait of Juan de Fuca. Nearshore areas along the Strait are considered to be a major migratory corridor.	Elwha Dungeness
Hood Canal The east face of the Olympic mountain range and small streams along the western Kitsap Peninsula drain into this distinct estuary.	Skokomish Mid Hood Canal (incl. Dosewallips, Duckabush and Hamma Hamma)
Whidbey Basin The Whidbey basin is the main estuarine area for the major Chinook-producing rivers in Puget Sound, and the migratory crossroads for most Puget Sound populations.	Skykomish Snoqualmie North and South Fork Stillaguamish Upper and Lower Skagit Upper and Lower Sauk Suiattle Cascade
Central/South Basin These basins were combined into a single geographic unit largely to reflect correlated risks from volcanic activity and urban-related effects.	Cedar River North Lake Washington Green/Duwamish Puyallup White Nisqually

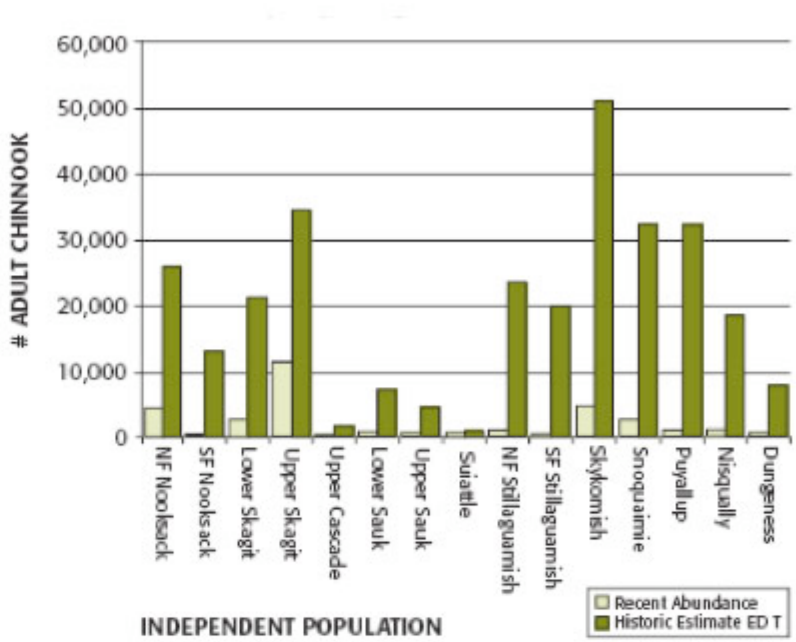


Figure 2. Comparison of recent (2000-2004) geometric mean of naturally spawning Puget Sound Chinook populations to estimates of historic capacity of in some Puget Sound watersheds using Ecosystem Diagnostic and Treatment (EDT) habitat models (Reprinted from NOAA Salmon 2007; courtesy of NOAA Fisheries).

Hood Canal Summer Chum

The summer run of Hood Canal chum salmon was listed as Threatened in 1999. A primary factor in this designation was the recognition that this stock comprises an important and distinct life history strategy within the species (NOAA 2007). Of the 16 historic spawning stocks of Hood Canal summer chum, eight are extant (NOAA 2007)(Table 2). In a recent review of this Threatened Evolutionarily Significant Unit (ESU), two genetically distinct populations were identified: a Strait of Juan de Fuca population (which includes the extant spawning aggregations Jimmycomelately, Snow, Salmon and Chimacum creeks) and a Hood Canal population (which includes the extant spawning aggregations Big and Little Quilcene, Dosewallips, Duckabush, Hamma Hamma, Union and Lilliwaup watershds)(Sands et al. 2009)(Figure 3). Maintaining diversity within and between these newly two newly identified populations will now be incorporated into the recovery goals for Hood Canal Summer Chum (Sands et al. 2009).

Table 2. Current (extant) and extinct populations of Hood Canal summer chum and supplementation/reintroduction programs (NOAA Salmon Recovery Plan 2007).

Population	Status	Supplementation/Reintroduction Program
Union River	Extant	Supplementation program began in 2000
Lilliwaup Creek	Extant	Supplementation program began in 1992
Hamma Hamma River	Extant	Supplementation program began in 1997
Duckabush River	Extant	—
Dosewallips River	Extant	—
Big/Little Quilcene River	Extant	Supplementation program began in 1992
Snow/Salmon Creeks	Extant	Supp. program began in 1992 in Salmon
Jimmycomelately Creek	Extant	Supplementation program began in 1999
Dungeness River	Unknown	—
Big Beef Creek	Extinct	Reintroduction program began in 1996
Anderson Creek	Extinct	—
Dewatto Creek	Extinct	Natural re-colonization occurring
Tahuya River	Extinct	—
Skokomish River	Extinct	—
Finch Creek	Extinct	—
Chimacum Creek	Extinct	Reintroduction program

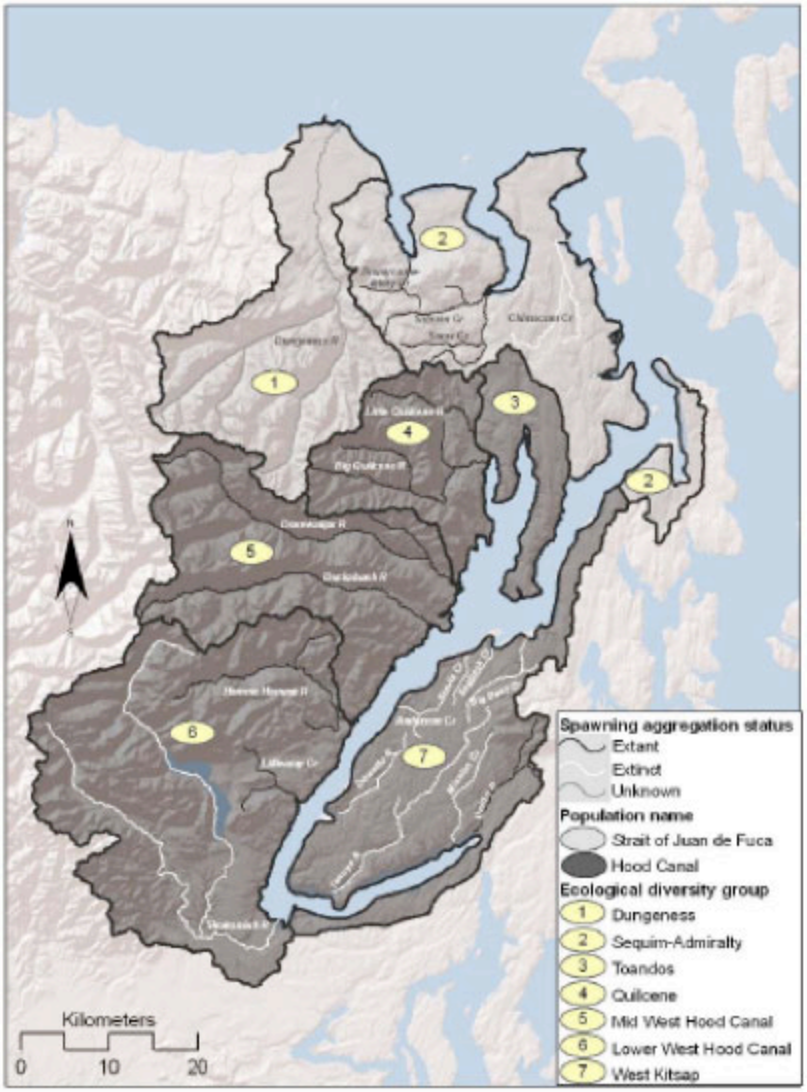


Figure 3. The two populations of the Hood Canal Summer Chum salmon ESU, including streams with spawning aggregations and seven ecological diversity groups (Reprinted from Sands et al. 2009; courtesy of NOAA Fisheries).

Steelhead

Steelhead in the Puget Sound ESU were federally listed as Threatened in 2007 (Hard et al. 2007). WDFW currently lists 53 spawning populations of steelhead in Puget Sound, the majority of which return in the winter to spawn. Less well studied and less abundant, the remaining populations return in the summer and are typically found in the northern portions of Puget Sound (Hard et al. 2007). The two largest populations of winter steelhead are also in the northern part of the sound, in the Skagit and Snohomish rivers (Hard et al. 2007)(Table 3).

Table 3. Geometric mean estimates of escapements of Puget Sound populations for all years of data (from ca. 1980 – 2004 for most populations) and for the 5 most recent years (2000 – 2004).

Estimates are based on hatchery and natural spawner (H+N columns) or only on natural spawners (N columns). Hatchery fish are not included in the Puget Sound ESU. NPS = Northern Puget Sound, SPS = Southern Puget Sound, HC = Hood Canal, SJF = Strait of Juan de Fuca, SSH = summer run steelhead, WSH = winter run steelhead, N/A = data not available (Hard et al. 2007).

Region	Run type	Population	H+N, all years	H+N, 5 years	N, all years	N, 5 years
NPS	SSH	Canyon	N/A	N/A	N/A	N/A
NPS	SSH	Skagit	N/A	N/A	N/A	N/A
NPS	SSH	Snohomish	N/A	N/A	N/A	N/A
NPS	SSH	Stillaguamish	N/A	N/A	N/A	N/A
NPS	WSH	Canyon	N/A	N/A	N/A	N/A
NPS	WSH	Dakota	N/A	N/A	N/A	N/A
NPS	WSH	Nooksack	N/A	N/A	N/A	N/A
NPS	WSH	Samish	684.2	852.2	500.8	852.2
NPS	WSH	Skagit	7,720.4	5,608.5	6,993.9	5,418.8
NPS	WSH	Snohomish	5,283.0	3,230.1	5,283.0	3,230.1
NPS	WSH	Stillaguamish	1,027.7	550.2	1,027.7	550.2
NPS	SSH	Tolt	129.2	119.0	129.2	119.0
SPS	SSH	Green	N/A	N/A	N/A	N/A
SPS	WSH	Cedar	137.9	36.8	137.9	36.8
SPS	WSH	Green	2,050.6	1,625.5	1,802.1	1,619.7
SPS	WSH	Lk. Washington	247.1	36.8	308.0	36.8
SPS	WSH	Nisqually	1,136.7	392.4	1,115.9	392.4
SPS	WSH	Puyallup	1,881.5	1,001.0	1,714.4	907.3
HC	WSH	Dewatto	27.0	24.7	24.0	24.7
HC	WSH	Dosewallips	70.6	76.7	70.6	76.7
HC	WSH	Duckabush	16.6	17.7	16.6	17.7
HC	WSH	Hamma Hamma	29.6	51.9	29.6	51.9
HC	WSH	Quilcene	16.8	15.1	16.8	15.1
HC	WSH	Skokomish	439.3	202.8	439.3	202.8
HC	WSH	Tahuya	131.8	117.0	113.9	117.0
HC	WSH	Union	57.1	55.3	55.0	55.3
SJF	SSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	Dungeness	311.2	173.8	311.2	173.8
SJF	WSH	Elwha	459.5	210.0	N/A	N/A
SJF	WSH	McDonald	N/A	N/A	149.8	96.1
SJF	WSH	Morse	132.6	103.0	105.8	103.0

Bull trout

Bull trout in Washington, including the Puget Sound Distinct Population Segment (DPS), were also listed as Threatened in 1999. The US Fish and Wildlife Service conducted an analysis of vulnerability to stochastic events across the spawning stocks of bull trout in Puget Sound, finding the Snohomish/Skyhomish, the Stillaguamish, and the Chester Morse Lake spawning stocks to be at the greatest risk (NOAA 2007)(Table 4).

Table 4. Bull trout risk levels for watersheds in Puget Sound (USFWS data)(NOAA Salmon Recovery Plan 2007)

Core Areas	Local and Potential Local Populations	Information on Abundance, Trends and Distribution	Risk from Stochastic Events
Chilliwack	Little Chilliwack River Upper Chilliwack River Selesia Creek (British Columbia & US) Depot Creek (BC & US) Airplane Creek (BC) Borden Creek (BC) Centre Creek (BC) Foley Creek (BC) Nesakwatch Creek (BC) Paleface Creek (BC)	Chilliwack Lake is an important source of rearing and forage for most local populations.	Intermediate risk if only the US populations are considered. Diminished risk if both US and Canadian populations are considered.
Nooksack	Lower Canyon Creek Glacier Creek Lower Middle Fork Nooksack R Upper MF Nooksack River Lower North Fork Nooksack R Middle NF Nooksack River Upper NF Nooksack River Upper South Fork Nooksack R Lower SF Nooksack River Wanlick Creek	Spawning occurs in all three forks of the Nooksack River and its tributaries. Fewer than 1000 spawners; most local populations have less than 100 adults.	Intermediate Risk
Lower Skagit	Bacon Creek Baker Lake Buck Creek Cascade River South Fork Cascade River Downey Creek Goodell Creek Illabot Creek Lime Creek Milk Creek Newhalem Creek Forks of Sauk River Upper South Fork Sauk River Straight Creek Upper Suiattle River Sulphus Creek Tenas Creek Lower White Chuck River Upper White Chuck River Sulphur Creek -Lake Shannon (potential local population) Stetattle Creek-Gorge Lake (potential local population)	Bull trout are known to spawn and rear in at least 19 streams/ stream complexes. This core area supports a spawning population of migrating bull trout numbering in the thousands. Connectivity and diversity of habitats are excellent except portions modified by dams. High abundance of pink salmon for forage.	Diminished Risk
Upper Skagit	Big Beaver Creek Little Beaver Creek Lightning Creek Panther Creek Pierce Creek Ruby Creek Silver Creek Thunder Creek (Diablo Lake) Deer Creek (Diablo Lake) (potential local population) Skagit River (BC) East Fork Skagit River (BC) Klesilkwa River (BC) Nepopekum Creek (BC) Skaist River (BC) Sumallo River (BC)	Populations are well distributed. British Columbia portion presumed healthy; status is generally unknown. 2 areas of concern due to lack of connectivity: Diablo Lake and Gorge Lake.	Intermediate risk if only the US populations are considered. Diminished risk if both US and Canadian populations are considered.
Stillaguamish	Upper Deer Creek South Fork Canyon Creek North Fork Stillaguamish River South Fork Stillaguamish River	Few known spawning areas. Fewer than 1000 spawners; most local populations have less than 100 adults. Snorkel surveys have found greater than 100 adults in the North Fork Stillaguamish R.	Increased risk
Snohomish-Skykomish	 South Fork Skykomish River Salmon Creek Troublesome Creek (primarily a resident population)	number of adult spawners is 500-1000. System has no lakes. Large portion of migratory segment are anadromous. North Fork Sky considered healthy by WDFW with 470-650 individuals on average, based on redd counts. South Fork Sky considered healthy by WDFW due to increasing numbers, and recolonization is occurring.	Increased risk
Chester Morse Lake	Boulder Creek Upper Cedar River Rex River Rack Creek Shotgun Creek (potential local population)	Area has few known spawning areas. Surveys in 2000-2002 documented 236-504 redds, with estimated 500-1000 spawners. Upper Cedar River and Rex River are the primary local populations in this core area. Upper Cedar River is the only known self-sustaining population in the Lake WA basin.	Increased risk
Puyallup	Carbon River Greenwater River Upper Puyallup and Mowich Rivers Upper White River West Fork White River Clearwater River (potential local population)	Fewer than 1000 spawners; most local populations have less than 100 adults. Known spawning areas are few and not widespread. Area has a low number of local populations. Portions within the National Park and wilderness area provide pristine habitat.	Intermediate risk

Trends

Chinook salmon

An analysis of 5-year population growth trends for Chinook salmon from 1986 - 2004 was conducted by NOAA fisheries. Of those populations that had been declining from 1986 – 1990, many exhibited positive growth over 1994 – 1998. (NOAA 2007) (Table 5). While productivity was not calculated for the most recent time period (2000-2004), the population means for this period were, in many cases, higher than that observed previously (NOAA 2007)(Table 5). Despite this positive trend, many populations remained low, including the Dungeness River and Skokomish spawning stocks (NOAA 2007)(Table 5).

Table 5. Geometric mean (5 year periods) of spawning abundances, hatchery contribution and productivity (number of return spawners per parent spawner) in Puget Sound Chinook Populations (NOAA Salmon Recovery Plan 2007).

	1986-1990			1994-1998			2000-2004	
Populations	Geometric Mean	% Hatchery Contribution	Productivity	Geometric Mean	% Hatchery Contribution	Productivity	Geometric Mean	% Hatchery Contribution
North + Middle Fork Nooksack	140	21%	1.29	263	67%	0.45	4,232	94%
South Fork Nooksack	243	7%	0.60	181	35%	1.20	303	46%
Lower Skagit	2,732	1%	0.59	974	1%	3.15	2,597	2%
Upper Skagit	8,020	2%	0.69	6,388	1%	1.60	12,116	4%
Upper Cascade	226	0%	0.88	241	0%	1.34	355	1%
Lower Sauk	888	0%	0.61	330	0%	2.35	825	0%
Upper Sauk	720	0%	0.57	245	0%	1.35	413	0%
Suiattle	687	0%	0.40	365	0%	1.20	409	0%
North Fork Stillaguamish	699	0%	0.92	862	35%	0.94	1,176	31%
South Fork Stillaguamish	257	0%	1.31	246	0%	1.22	205	0%
Skykomish	3,204	14%	0.52	3,172	52%	0.82	4,759	39%
Snoqualmie	907	12%	1.23	1,012	33%	1.68	2,446	14%
Sammamish	388	41%	0.28	145	74%	2.72	243	69%
Cedar	733	9%	0.51	391	17%	0.97	412	21%
Green/Duwamish	7,966	62%	0.50	7,060	71%	1.00	13,172	34%
White	73	56%	7.51	452	82%	1.49	1,417	28%
Puyallup	1,509	15%	1.86	1,657	40%	0.67	1,353	31%
Nisqually	602	3%	4.22	753	21%	1.38	1,295	25%
Skokomish	1,630	69%	0.48	866	69%	0.34	1,479	80%
Mid Hood Canal	87	26%	1.41	182	26%	1.31	202	46%
Dungeness	185	83%	0.12	101	83%	0.70	532	83%
Elwha Nat Spawners	2,055	34%	0.46	512	61%	1.03	847	54%
Elwha Nat+Hat Spawners	3,887	34%	0.67	1,679	61%	1.27	2,384	54%

Hood Canal Chum salmon

Population growth rates for Hood Canal summer chum salmon were all positive over short- time frames (1999-2002), but only two of the eight spawning aggregations (Union River and Big/Little Quilcene River) displayed positive growth rates over longer time scales (1970s – 2002) (Table 6). The latter two are both constituents of the Hood Canal genetically independent population (Sands et al. 2009), and experienced declines in the 1980- 1990s followed by recent increases (Sands et al. 2009)(Figure 4).

Table 6. Mean abundance of Hood Canal summer chum in each watershed and long-term (1970s – 2002) and short-term (1999 - 2002) population growth trends (NOAA Salmon Recovery Plan 2007).

Population	Geometric mean escapement (1999-2002)	Long Term Trend (a value of 1.0 indicates that the population is replacing itself)	Short Term Trend
Union River	594	1.08	1.10
Lilliwaup Creek	13	0.88	1.00*
Hamma Hamma River	558	0.90	1.20
Duckabush River	382	0.91	1.14
Dosewallips River	919	0.96	1.25
Big/Little Quilcene River	4,512	1.05	1.62
Snow/Salmon Creeks	1,521	0.99	1.24
Jimmycomelately Creek	10	0.88	0.82*
* Supplementation programs at Jimmycomelately and Lilliwaup reduced the number of spawners released to achieve escapement naturally.			

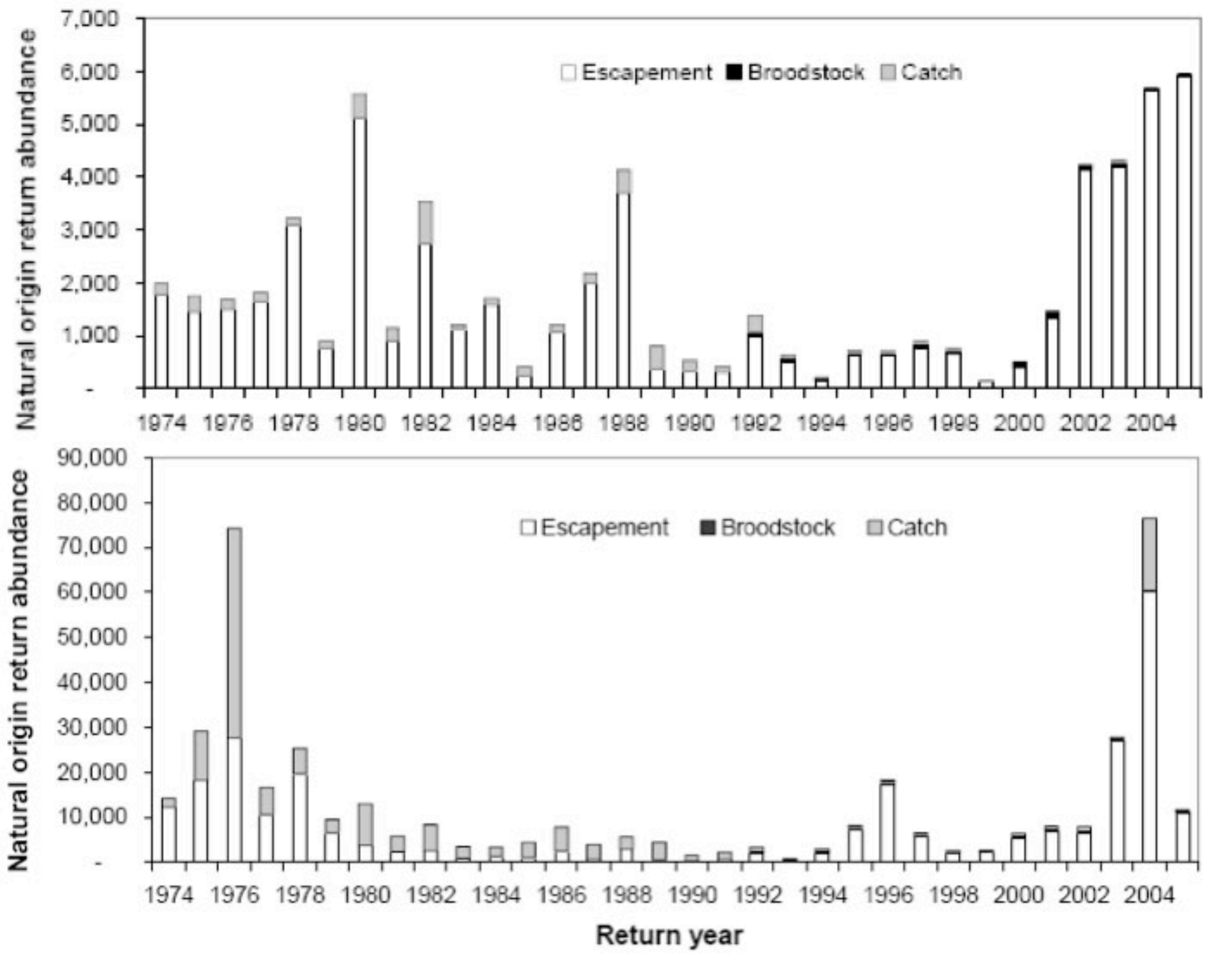


Figure 4. Annual return abundances of natural-origin summer chum salmon of the Strait of Juan de Fuca region (TOP) and the Hood Canal region (BOTTOM) from 1974 – 2005 (Reprinted from Sands et al. 2009; courtesy of NOAA Fisheries).

Steelhead Analyses utilizing all years of available data (ca. 1980 – 2004) and the 10 most recent years (1995-2004) indicated that most Puget Sound steelhead populations exhibited significantly declining trends in natural escapements, particularly in the southern Puget Sound (e.g., the Cedar, Lake Washington, Nisqually and Puyallup winter run populations) (Hard et al. 2007)(Table 7). Increasing populations were observed in the Samish and Hamma Hamma winter run populations (Hard et al. 2007)(Table 7).

Table 7. Estimates of temporal trends in escapement (E) and total run size(R) (log-transformed) for naturally produced Puget Sound. Positive values indicate a growing population, negative values indicate a declining one. Asterices indicate level of significance (Hard et al. 2007).

Region	Run type	Population	E, all years	E, 10 years	R, all years	R, 10 years
NPS[a]	SSH[b]	Canyon	N/A[c]	N/A	N/A	N/A
NPS	SSH	Skagit	N/A	N/A	N/A	N/A
NPS	SSH	Snohomish	N/A	N/A	N/A	N/A
NPS	SSH	Stillaguamish	N/A	N/A	N/A	N/A
NPS	WSH[d]	Canyon	N/A	N/A	N/A	N/A
NPS	WSH	Dakota	N/A	N/A	N/A	N/A
NPS	WSH	Nooksack	N/A	N/A	N/A	N/A
NPS	WSH	Samish	+0.067**	+0.061**	+0.019	+0.014
NPS	WSH	Skagit	−0.002	−0.010	−0.021	−0.056
NPS	WSH	Snohomish	−0.019	+0.035*	−0.086	N/A
NPS	WSH	Stillaguamish	−0.065****	N/A	−0.110*	N/A
NPS	SSH	Tolt	+0.025	+0.034	−0.107	−0.021
SPS[e]	SSH	Green	N/A	N/A	N/A	N/A
SPS	WSH	Cedar	−0.179**	N/A	−0.299*	N/A
SPS	WSH	Green	+0.008	−0.016**	−0.048	−0.069*
SPS	WSH	Lk. Washington	−0.180****	−0.215****	−0.300*	−0.274
SPS	WSH	Nisqually	−0.084****	−0.147****	−0.097	−0.159**
SPS	WSH	Puyallup	−0.062****	−0.074****	−0.103**	−0.103**
HC[f]	WSH	Dewatto	N/A	N/A	N/A	N/A
HC	WSH	Dosewallips	N/A	N/A	N/A	N/A
HC	WSH	Duckabush	+0.017	−0.018	+0.017	−0.019
HC	WSH	Hamma Hamma	+0.291*	+0.264	+0.291*	+0.264
HC	WSH	Quilcene	−0.006	N/A	−0.006	N/A
HC	WSH	Skokomish	−0.075****	−0.136**	−0.109*	−0.136**
HC	WSH	Tahuya	+0.009	−0.002	+0.004	−0.021
HC	WSH	Union	+0.008	+0.002	+0.008	+0.002
SJF[g]	SSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	Dungeness	−0.076****	−0.093**	−0.083	−0.093
SJF	WSH	Elwha	N/A	N/A	N/A	N/A
SJF	WSH	McDonald	−0.031	+0.009	−0.362**	−0.221*
SJF	WSH	Morse	−0.006	−0.015	−0.030	−0.050

[a] NPS = Northern Puget Sound.
[b] SSH = Summer run steelhead.
[c] N/A = Data not available.
[d] WSH = Winter run steelhead.
[e] SPS = Southern Puget Sound.
[f] HC = Hood Canal.
[g] SJF = Strait of Juan de Fuca.

Bull trout

There is a paucity of reported data on the population trends of bull trout in Puget Sound.

Uncertainties

Because of the wide array of life history types exhibited and habitats utilized by salmonids, the list of human threats as well as environmental and ecological drivers of salmonid abundance is long. These include hydropower, harvest, reduction in freshwater habitat quality and quantity, interactions with other fish, birds and marine mammals, ocean conditions and negative impacts of hatchery-reared salmon (Ruckelshaus et al. 2002). These drivers likely apply to both listed and non-listed salmonids in Puget Sound.

Summary

Salmon and trout are key ecological, cultural and economic components of the Puget Sound ecosystem. The number of Chinook salmon has increased since being listed in 1999, although population numbers remain well below target abundances. Hood Canal Summer chum salmon populations have shown some increases since their listing. Population abundance data for the two listed trout and charr species have not been published in citable reports or other publications.

Literature Cited

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Fresh, K. L. 2006. Juvenile Pacific salmon and the nearshore ecosystem of Puget Sound., Published by Seattle Districut, U.S Army Corps. of Engineeers, Seattle, WA.

Hard, J. J., J. M. Myers, M. J. Ford, R. G. Cope, G. R. Pess, R. S. Waples, G. A. Winans, B. A. Berejikian, F. W. Waknitz, P. B. Adams, P. A. Bisson, D. E. Campton, and R. R. Reisenbichler. 2007. Status review of Puget Sound steelhead (*Oncorhynchus mykiss*). U.S. Department of Commerce, NOAA Tech Memo, Seattle, WA.

NOAA. 2007. Puget Sound Salmon Recovery Plan. Shared Strategy for Puget Sound and NOAA Fisheries, Seattle, WA.

O'Neill, S. M., and J. E. West. 2009. Marine distribution, life history traits, and the accumulation of polychlorinated biphenyls in chinook salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138:616-632.

Quinn, T. P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society, Bethesda, Md.

Quinn, T. P., B. R. Dickerson, and L. A. Vøllestad. 2005. Marine survival and distribution patterns of two Puget Sound hatchery populations of coho (*Oncorhynchus kisutch*) and chinook (*Oncorhynchus tshawytscha*) salmon. Fisheries Research 76:209-220.

Ruckelshaus, M. H., P. Levin, J. B. Johnson, and P. M. Kareiva. 2002. The Pacific salmon wars: What science brings to the challenge of recovering species. Annual Review of Ecology and Systematics 33:665-706.

Sands, N. J., K. Rawson, K. P. Currens, W. H. Graeber, M. H. Ruckelshaus, R. R. Fuerstenberg, and J. B. Scott. 2009. Determination of independent populations and viability criteria for the Hood Canal summer chum salmon evolutionarily significant unit. U.S. Department of Commerce, NOAA Fisheries, Seattle, WA.

WDFW. 2002. 2002 SaSI Introduction. Washington Department of Fish and Wildlife, Olympia, WA.

Marine birds

Background

Puget Sound is important for nesting, wintering, and migration of numerous bird species associated with the marine environment. More than 70 bird species regularly utilize Puget Sound during some or all stages of their life histories (Buchanan 2006), but only a portion of these are actively being investigated. Studies have focused primarily on abundance and distribution, habitat utilization, foraging behavior, and contamination levels.

Multispecies comparisons

Information pertaining to marine bird distribution and abundance prior to the 1970s resides primarily in anecdotal accounts (Rathbun 1915, Jewett 1953) and systematic surveys held during Christmas Bird Counts (CBCs), which became consistent and widespread in the 1960s. Since the 1970s, the most comprehensive census of marine birds in northern Puget Sound was conducted as part of the Marine Ecosystems Analysis (MESA) program of 1978-1979 (Wahl 1981). The MESA study was a large-scale survey jointly funded by the Department of Commerce (DOC) and the Environmental Protection Agency (EPA) as a response to oil spill threats in the Strait of Juan de Fuca. It included aerial, land-based, and ferry-based transect surveys north of Admiralty Inlet, within portions of the Straits of Juan de Fuca and Georgia, and the Canadian Gulf Islands. Notably, the study included only the southernmost portion of the Strait of Georgia and not Puget Sound itself.

Beginning in 1992, the Puget Sound Ambient Monitoring Program (PSAMP) began collecting observations of marine birds in the non-breeding season; this currently is the only source of continuous multi-species monitoring of marine birds in Puget Sound. The annual surveys consist of aerial transects covering nearshore habitat and offshore habitat/open waters throughout Puget Sound and the southern shore of the Strait of Juan de Fuca (Figure 1). Aircraft-based observers record all bird species seen below the high tide line, but monitoring goals and data summaries emphasize certain alcid, diving duck, loon, and grebe species.

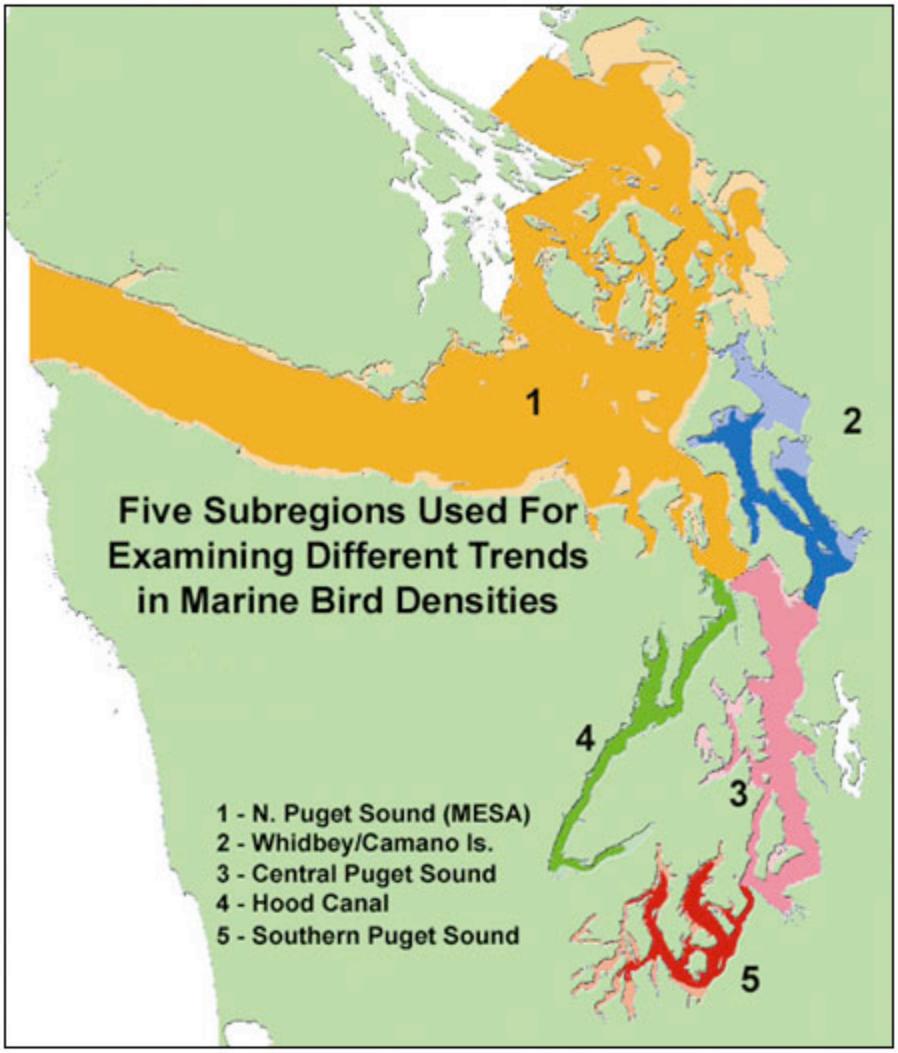


Figure 1. Map of PSAMP subregions (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Nysewander et al. (2005) evaluated long-term changes in abundance in several species of marine birds by comparing the PSAMP results from 54 aerial transects with results from nearly identical MESA transects. Results of this analysis revealed significant declines in 13 of the 20 species or species groups studied, including declines in at least one species from each marine bird family found in northwestern Washington. For some species, such as the western grebe (*Aechmophorus occidentalis*) and long-tailed duck (*Clangula hyemalis*), declines were as high as 95% and 91%, respectively. Although methodologies used in MESA and PSAMP surveys were relatively comparable, differences did exist, for example in the locations and habitat types surveyed by the aerial methods, and in the proportion of the MESA baseline data that was from aerial, land-based and ferry-based surveys. Furthermore, the PSAMP used aerial surveys, but the potential bias associated with avoidance of aircraft by birds has not been evaluated.

Results from Nysewander et al. (2005) and other studies (e.g. Wahl 2002) sparked concern over declines in marine birds in Puget Sound. In acknowledgment of these concerns and the multiple problems associated with comparing results across disparate survey methodologies, the Western Washington University (WWU) or WWU/MESA comparison study was initiated (Bower 2009). The WWU/MESA comparison study replicated land-based and ferry-based transect portions of the MESA surveys over two winters (2003-2004 and 2004-2005). Results of the WWU/MESA comparison of data were largely consistent with the MESA/PSAMP comparison (Nysewander et al. (2005), although a few results diverged. To perform a third comparison of marine bird observations over time, Bower (2009) analyzed annual Christmas Bird Count (CBC) data from 11 count circles north of Puget Sound (1975-1984 and 1998-2007). Table 1 summarizes characteristics of the data sets used by to make comparisons (Bower (2009).

Table 1. Attributes of bird surveys compared in Bower (2009)

Study	Year(s)	Geographic area	Methods
Marine Ecosystems Analysis (MESA)	Jan-Dec 1978-1979	Admiralty Inlet (S), Tsawwassen-Schwartz Bay BC Ferry (N), Neah Bay (W), and WA mainland (E)	Shore-based point counts, ferry and small boat transects, aerial transects
Puget Sound Ambient Monitoring Program (PSAMP)	Winter 1992-1999	Straight coastline between Admiralty Inlet (S), Strait of Georgia (N), Neah Bay (W), and WA mainland (E)	Aerial transects compared with 1970s MESA aerial transects
Western Washington University (WWU)	Sept-May 2003-2004 and Sept-May 2004-2005	Admiralty Inlet (S), Tsawwassen-Schwartz Bay BC Ferry (N), Sand Juan Islands (W), WA mainland (E)	Shore-based point counts and ferry transects compared with 1970s MESA shore-based point counts and ferry transects
Christmas Bird Count (CBC)	1975-1984 and 1998-2007	Salish Sea, including 8 BC and 3 WA CBC circles	Standard CBC methods for 11 CBC circles, data from 1975-1984 with data from 1998-2007

Bower (2009) reported a 29% decline in the total number of marine birds in inland waters of the Salish Sea – which includes areas and data from outside the Puget Sound Basin – between 1978/79 and 2003-2005 (Figure 2). It should be noted, however, that this overall decline can be substantially influenced by changes exhibited by certain individual species. For example, of the 37 most common overwintering marine species, 14 showed significant declines and six showed significant increases. Notably, the largest declines were observed among taxonomically diverse groups, including common murre (*Uria aalge*) (–92.4%), western grebe (–81.3%), surf scoter (*Melanitta perspicillata*) (–59.8%) and brant (*Branta benicla*) (–73.2%). Species that showed increases in abundance included double-crested cormorant (*Phalacrocorax auritus*) (+97.7%) and pigeon guillemot (*Cepphus columba*) (+108.9%). Results from the CBC data comparison revealed significant declines in seven of the 37 most common species or species groups, with significant increases in three species (Bower 2009).

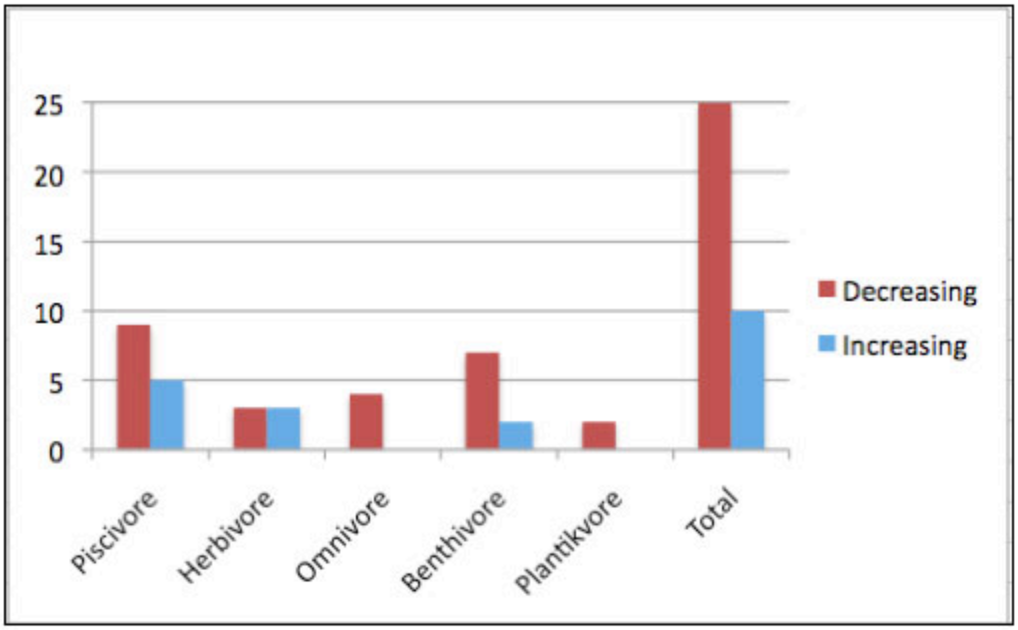


Figure 2. Changes in mean abundance among feeding guilds for 35 common overwintering marine birds in the Salish Sea between 1978/79 and 2003-2005 (data from Bower 2009)

Seventeen species or species groups were common to all three studies (the WWU/MESA comparison, the PSAMP/MESA comparison, and the CBC data comparison (Bower 2009). The PSAMP/MESA comparison revealed declines in more species (14 of 17) than did either the WWU/MESA comparison (six of 17) or the CBC comparison (three of 17) (Table 2). The PSAMP/MESA comparison showed no change or an increase in just three of 17 species or species groups, whereas the WWU/MESA comparison found no change or an increase in six of 17 and the CBC comparison found no change or an increase in eight of 17 species or species groups. Despite these differences, several consistencies emerge. First, the number of species declining exceeded the number of species increasing in all three comparisons. Second, three taxa -- western grebe, all scaup, and marbled murrelet-- showed declines across all three studies. And finally, only two species (Harlequin Duck, Bald Eagle) showed significant increases across all the three comparisons (Bower 2009).

Table 2. Comparison of percent change detected in three studies of non-breeding marine bird abundance for 17 species or species groups in Puget Sound (Bower 2009)

	Studies		
Species	PSAMP/MESA	WWU/MESA	Recent/historic CBCs

Common Loon	-64a		+49a		+13	
All loons	-79a			-33		-47
Red-necked Grebe	-89a		-46a			-35
Horned Grebe	-82a		-72a			-30
Western Grebe	-95a		-81a		-86a	
Double-crested Cormorant	-62a		+98a		+171a	
All cormorants	-53a		-8.3a			-25
Great Blue Heron		-19	+51			-16
Brant	-66a			-73	+1027a	
All scaup	-72a		-65a		-51a	
Harlequin Duck	+189a		+20		+7	
Long-tailed Duck	-91a			-44	+49	
All scoters	-57a		-33a			-8
Bufflehead	+20			-11	+5	
Bald Eagle	+35		+187a		+28	
Pigeon Guillemot	-55a		+109a		+15	
Marbled Murrelet	-96a		-71a		-69a	

a Statistically significant

In summary, widespread changes in the abundance of marine birds during the non-breeding season have occurred over the last 30 years in the Salish Sea (Nysewander et al. 2005, Bower 2009). Causes of these declines are not adequately known.

Scoters

Puget Sound supports some of the largest wintering scoter populations on the west coast of North America (Wahl 1981), where they feed on regionally-abundant bivalves and forage fish roe. Puget Sound is also one of the three most important staging areas and one of two major molting areas for other west coast scoter populations, including scoters that winter in California, Mexico, and British Columbia. Scoter populations in Puget Sound, including the wintering, staging, and molting populations, consist primarily of surf scoters and white-winged scoters (*M. fusca*). Black scoters (*M. perspicillata*) are also present, but in much smaller numbers. Surf scoters are one of the most abundant diving ducks in Puget Sound between September and May, with the highest densities in southern and central Puget Sound (Nysewander et al. 2005). Washington’s wintering scoters spend eight to 10 months in marine waters, with males spending approximately a month longer than females, before migrating to the Canadian interior to breed on freshwater lakes.

Scoters in Puget Sound are found most often in shallow coastal waters (< 20 meters depth) over a broad range of substrates, including pebbles, sand, mud, cobble, and rock. Previously thought to subsist on a relatively narrow diet of bivalves, scoters are now understood to adjust foraging patterns and locations to take advantage of ephemeral food sources. During much of the winter, they forage on newly-settled mussels and soft substrates inhabited by clams and other shellfish. In spring, some scoters in the region feed on herring eggs where available and flocks of surf scoters regularly track the northward progression of spawning events to consume this abundant and energy-rich source of food (Vermeer 1981). Anderson et al. (2008) found that prey such as crustaceans and polychaetes associated with eelgrass habitats comprise a substantial part of scoter diets in late summer.

Scoters observed in both nearshore and offshore waters during PSAMP winter monitoring efforts between 1992 and 2008 ranged in mean overall densities from 9.2 to 19.4 birds per km2 (Evenson et al. 2010). The density indices reported for nearshore areas, which scoters favor, ranged from 34.8 to 70.4 birds per km2. Figure 3 shows scoter densities between 1992 and 2008. Of all scoters counted along transects sampled during 1992-2008 winter surveys, between 33% and 90% were identified to species in any single year. Of these, surf scoters comprised 55-82%, white-winged scoters comprised 17-40%, and black scoters made up 0.2-9%. WDFW currently is conducting species/age/sex ratio surveys by boat to provide a better estimate of species proportions (Evenson 2010).

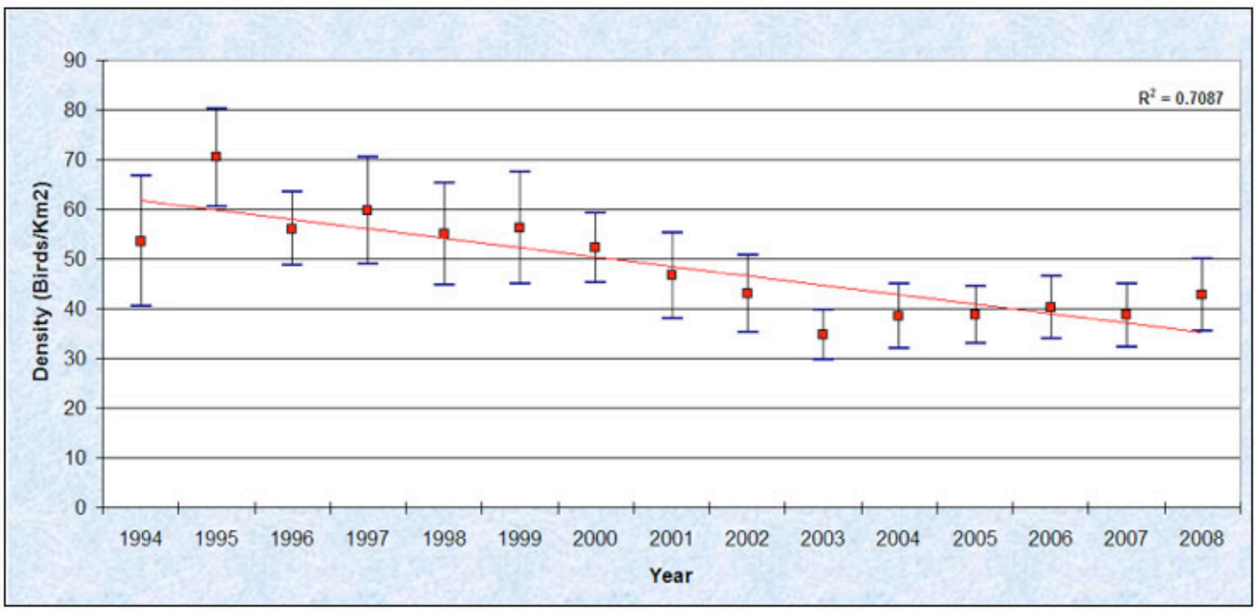


Figure 3. Mean winter densities of scoters in nearshore (<20 m) habitats of the inner marine waters of Washington state, 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Bower (2009) demonstrated that as a group scoters showed significant declines in both the PSAMP/MESA (-57%) and WWU/MESA (-33%) comparative studies. Surf scoters declined by 60% in the WWU/MESA comparison; however, nearly half of this decline is attributed to the collapse of the Cherry Point herring stock that occurred between the two survey periods (Stout 2001, Bower 2009). The evidence for this decline is compelling: more than 40,000 surf scoters were observed by MESA researchers in 1978 and less than 1,000 surf scoters were seen by WWU researchers at the same location in 2004 and 2005.

Comparisons of annual changes in density in the inner marine waters of Washington between 1992 and 2008 suggest that the scoters declined from the early 1990s through 2003, but that since 2003, densities have been relatively stable (Figure 3)(Evenson et al. 2010). However, spatial variation in rates of decline exist, for example in the Whidbey/Camano (North Puget Sound)area, where scoter densities have continued to decline (Figure 4)(Evenson et al. 2010). In 1993, the densities in the Whidbey/Camano area were the highest in the inner marine waters of Washington, but by 2008 densities in the Whidbey/Camano were the lowest (Figure 4)(Evenson et al. 2010).

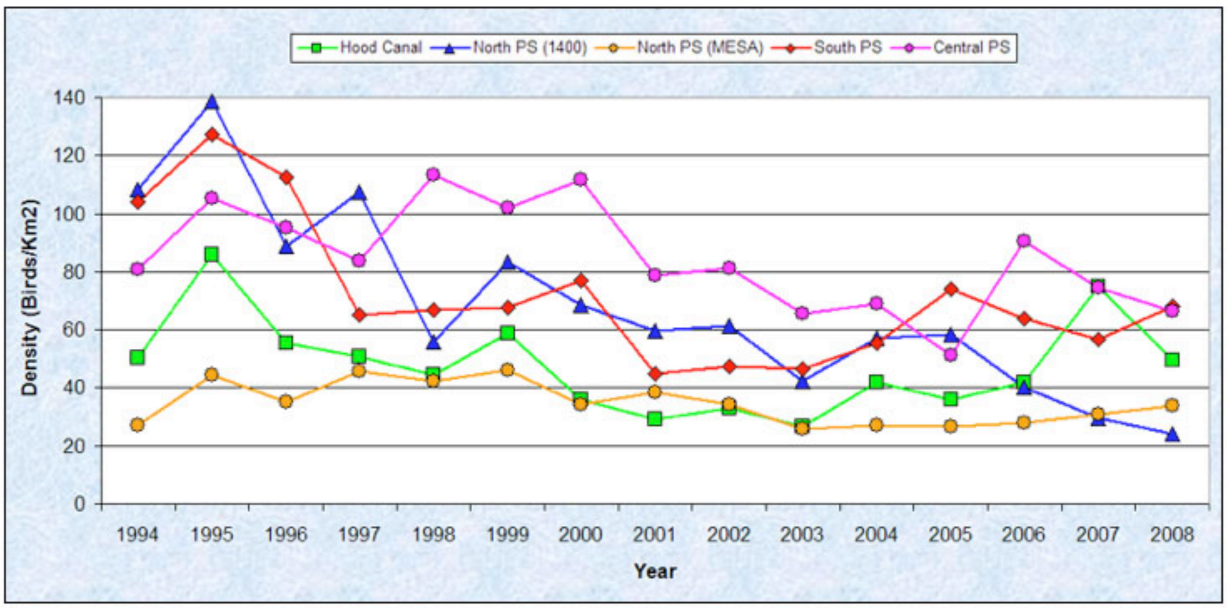


Figure 4. Comparison of winter scoter densities by region in the nearshore (<20 m) inner marine waters of Washington state (Puget Sound), 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Loons and Grebes

Several species of loons and grebes spend a substantial portion of the winter in Puget Sound where they utilize a variety of marine habitats. Loon species include the common loon (*Gavia immer*), Pacific loon (*G. pacifica*), and red-throated loon (*G. stellata*). Common loons are widespread and fairly common during winter in almost all nearshore marine habitats, and in most freshwater habitats, except rivers, typically occurring as single birds or in small numbers. They are rare breeders in Washington waters with the majority nesting throughout Canada and Alaska. Common loons were listed as sensitive by WDFW because they are a rare breeding species in the state and are vulnerable to a number of threats, including destruction or alteration of nesting habitat, poor water quality (i.e., degradation of lakes), and human activity (Richardson et al.

2000). Pacific loons are also widespread and common during winter, but occur further offshore than common loons and are more likely to congregate. Flocks of Pacific loons feed on schools of small fish near banks, tidal rips, and other hydrographic features of deeper waters. This species breeds in eastern Siberia and from northern coastal Alaska across to Baffin Island and Hudson Bay in North America. Red-throated loons are widespread and fairly common during winter in Puget Sound; they breed throughout Alaska, Canada, Greenland, and northern Europe Asia, with the very southern portion of their range extending south to southern Vancouver Island. Although red-throated loons can frequent many different types of marine waters, they tend to favor estuaries and shallow offshore areas, aggregating at times in areas where prey species are concentrated by tidal conditions.

Western grebes utilize marine and fresh waters in Puget Sound between October and April and tend to occur in groups. The primary wintering habitat for the larger flocks of western grebes are in offshore (>20m depth) marine waters with minimal tidal current flow, where they prey on schooling forage fish, although they may also occur in many saltwater situations and on inland lakes. Western grebes gather in large resting groups during the daytime hours and then disperse at night to forage. Major concentration areas have been identified through PSAMP aerial surveys (Evenson et al. 2010). Western grebes breed from southern British Columbia and the prairie Provinces in Canada south to Mexico.

Comparisons of survey data (Nysewander et al. 2005, Bower 2009) reveal that Puget Sound loon and grebe species have declined significantly in recent decades. Bower (2009) detected declines in loons as a group in all three comparative studies as follows: 64% decline in MESA/PSAMP comparison; 33% decline in WWU/MESA comparison; and 47% decline in historic/recent CBC comparison (Table 2). Records from the annual PSAMP winter aerial surveys from 1992 to 2008 show that loons constituted 0.8% of all marine birds surveyed (Evenson et al. 2010). The majority of loons were identified to species (common loon [28%], Pacific loon [27.9%], and red-throated loon [32.5%]) and occurred in both nearshore and offshore waters.

Among the three loon species, densities were lowest in the common loon, ranging from 0.17 to 0.57 birds per km2. A comparative analysis of common loon densities reported in MESA (1978/79) and PSAMP (1992-1999) surveys showed a 64% decline (Nysewander et al. 2005). Conversely, Bower (2009) reported 49% and 13% increases as shown by WWU/MESA and the historic/recent CBC data comparisons, respectively, which include survey data through the mid-2000s. It is unclear whether these changes reflect some degree of recovery since 1999, shifts in distribution, or are an artifact of differing or variously effective methodologies (see Uncertainties section, below).

Densities of Pacific loons observed during PSAMP winter surveys ranged from 0.26 to 1.21 birds per km2 in 1994-2008, with higher densities (10 and 89 birds per km2) observed in areas where flocks concentrate. Pacific loon winter densities appeared to be relatively stable over the period 1994-2008, although this result may be confounded by other loon species (Evenson et al. 2010). A comparison between MESA and PSAMP data was not made for Pacific loons due to the difficulty of distinguishing Pacific loons from red-throated loons in aerial surveys. Analysis of PSAMP subregional density indices suggest that Pacific loons favor certain subregions, such as northern Puget Sound, Whidbey/Camano Islands, and Central Puget Sound near Bainbridge Island (Evenson et al. 2010).

Red-throated loon densities of 0.17 to 1.20 birds per km2 were observed during PSAMP winter surveys of nearshore and offshore areas between 1994-2008 (Evenson et al. 2010). Densities appear to have been relatively stable over the past two decades (Evenson et al. 2010), although this species is not clearly separated from other loon species in some survey data.

Grebes

All grebe species wintering in Washington marine waters have exhibited some degree of decline over the last two decades, but western grebes have declined most sharply (Evenson et al. 2010). Overall densities for western grebes, combined for both nearshore and offshore waters, ranged from 3.9 to 13.2 birds per km2, while densities in the vicinities of the flocks ranged from 50 to 1,343 birds per km2 (Figure 5). A comparative analysis of western grebe densities reported by MESA (1978/79) and PSAMP (1992-1999) surveys showed a 95% decline (Nysewander et al. 2005). Bower (2009) noted that declines were observed in all three comparative studies (Table 2), across of the Salish Sea, and in every month of the MESA/WWU comparative surveys. Density indices reported by PSAMP winter monitoring surveys between 1992-2008 suggest that this species is still declining (Evenson et al. 2010).

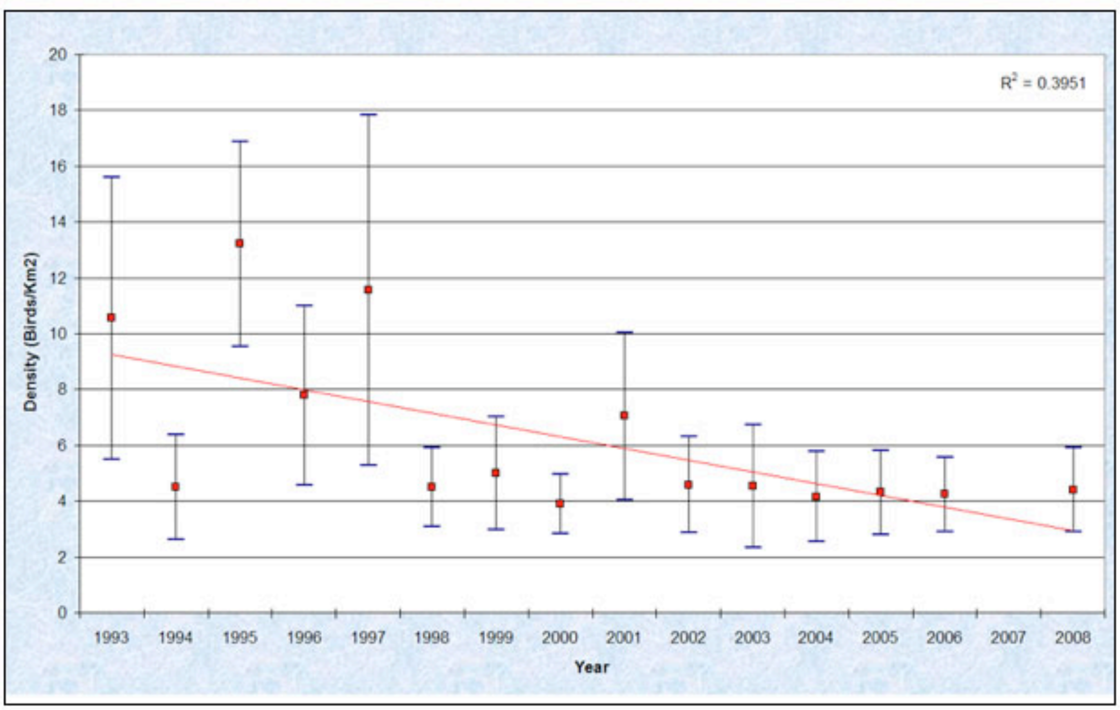


Figure 5. Winter Trends in Western Grebe Densities in the Inner Marine Waters of Washington State, 1993-2008 (Reprinted from Evenson et al. 2010 with permission from Washington Department of Fish and Wildlife)

Alcids

Several alcid species utilize marine waters of Puget Sound during winter months, with some species breeding along coastlines and on islands. Pigeon guillemots (*Cepphus columba*) are common and widespread residents that feed in nearshore habitats, along rocky shorelines, passes, banks, areas with tidal currents and rips, as well as in shallow embayments. These birds are seldom seen in flocks, except near colonies during breeding, although they can aggregate in productive feeding areas such as tidal convergences and passes. Pigeon guillemot nest in nearly every small-island or saltwater coastline habitat, with larger colonies are found in San Juan, Jefferson, Island, and Clallam Counties. Smaller colonies and single pairs are found throughout Puget Sound, making them the second most common breeding seabird in Puget Sound.

Rhinoceros auklets (*Cerorhinca monocerata*) are found throughout Puget Sound in both coastal habitats and far from land. Most often they often feed close to shore, especially where tidal currents near islands create localized upwelling and trophic intensification. Flocks may overnight in protected bays and forage farther out to sea during the day. Rhinoceros auklets in Washington nest at three main sites: Destruction Island, Protection Island, and Smith Island. Smaller numbers nest at a few other sites in Puget Sound.

Marbled murrelets (*Brachyramphus marmoratus*) are small, fast-flying seabirds present year round in coastal areas throughout Washington. They are non-colonial alcids that breed in mature inland forests up to 84 km from marine shorelines that support prey such as small schooling fish or invertebrates in shallow waters (Raphael 2006). Areas of winter concentration in Washington include the southern and eastern end of the Strait of Juan de Fuca, Sequim, Discovery and Chuckanut Bays, and the San Juan Archipelago. In 1992, the Pacific coast population of marbled murrelets south of the Canadian border was listed as Threatened by both USFWS and the State of Washington. Critical habitat in Washington, Oregon and California was designated in 1996. Primary threats to marbled murrelets include the loss and modification of nesting habitat, primarily due to commercial timber harvesting of older forests, effects resulting from oil spill pollution, and to a much lesser degree, risks associated with capture in commercial fisheries gear (Ralph et al. 1995).

In 2003, a WDFW survey of pigeon guillemot colonies in Puget Sound reported at least 471 colonies, representing approximately 16,000 breeding birds (Evenson et al. 2003). Long-term changes in pigeon guillemot populations are not known due to absence of historical data. Records from annual PSAMP aerial surveys show that pigeon guillemot densities were highest in nearshore habitats (<20m depth), where they ranged from 0.26 to 1.18 birds per km2 in 1992-2008 (Evenson et al. 2010). Densities from the inner marine waters of Washington during winters 1993-2008 increased from 1993-1997, and then remained stable through 2008. A comparative analysis of pigeon guillemot densities recorded by MESA (1978/79) and PSAMP (1992-1999) surveys showed a 56% decline over that period (Nysewander et al. 2005). However, Bower (2009) reported a 109% increase in pigeon guillemot density based on the WWU/MESA comparative study, which covered a slightly longer time period (Table 2). The inconsistencies likely reflect differences in sampling between the studies (Bower 2009) and a the lack of knowledge of pigeon guillemot post-breeding dispersal patterns (Evenson et al. 2010).

Rhinoceros auklet breeding populations in Puget Sound are concentrated on Protection Island and Smith Island. Protection Island hosts 70% of the breeding birds within Washington's inner marine waters (Speich et al. 1989). Estimates of the breeding population of Rhinoceros auklets on Protection Island have shown a 30% decline in breeding pairs with more than 17,000 breeding pairs in 1975 (Wilson and Manuwal 1986) decreasing to approximately 12,000 pairs in 2000 (Wilson 2005).

In 2006, marbled murrelet population size was estimated to be about 22,000 in Washington, Oregon, and California (Huff et al. 2006), compared with approximately 860,000 in Alaska and 55,000 to 78,000 in British Columbia in 2004 (McShane 2004). At-sea counts of marbled murrelets using boat-based observer transects were conducted from 2000 to 2009 as part of effectiveness monitoring of the Northwest Forest Plan. In 2009, USFWS conducted a five-year status review of the Northwest Forest Plan and determined that marbled murrelets in Puget Sound had continued to decline significantly since the previous review conducted in 2002 (Pearson et al. 2010). The population estimate for marbled murrelets in all zones in the Northwest Forest Plan area (Washington, Oregon and California) was 17,791 (95% confidence interval: 14,631 – 20,952). Estimates from the 9 years of monitoring have ranged from 17,354 to 23,673. The 2009 population estimate for Puget Sound and Juan de Fuca Strait east of Cape Flattery from at-sea surveys was 5,623 birds (95% confidence interval: 3,922 – 8,352 birds). The annual rate of decline for the 2001-2009 period was 7.0% (standard error = 1.8%; Pearson et al. 2010). For Washington State overall, there was a significant decline in murrelet density for the 2001-2009 period (Pearson et al. 2010). The largest concentrations of birds occurred in northern Puget Sound and the Strait of Juan de Fuca.

High Arctic Black Brant

High arctic black brant are a subpopulation of brant geese that utilize Puget Sound shallow bays and saltwater marshes from late November through May. They breed in the high arctic of western Canada, primarily on Melville Island and Prince Patrick Island, and then stage for over a month at Izembek Lagoon in Alaska before heading to wintering grounds in Puget Sound. Brant wintering habitats are usually characterized by an abundance of eelgrass and marine algae (e.g., Padilla, Samish and Fidalgo Bays in Skagit County). Large concentrations of brant may gather at Dungeness Spit and Willapa Bay, but smaller flocks are present in the southern Puget Sound. Because of their strong dependence on certain plants, fidelity to wintering and breeding locations, and because some populations live in harsh environments, brant are more vulnerable to periodic breeding failures and occasional heavy losses from starvation than are most other geese (Reed et al. 1998).

Results from the comparative MESA and PSAMP studies showed that brant abundance varied widely over spatial and temporal scales (Bower 2009). Brant exhibited declines in the PSAMP/MESA comparision (-66%) and WWU/MESA comparison (-73.2%), but increased by more than 1000% in the CBC comparison data (Bower 2009). The large decline in the WWU/MESA comparison was principally driven by a decline in numbers on the primary wintering grounds of Padilla and Samish Bay. Outside these two locations, brant numbers showed a slight increase. CBC comparison data showed increases in brant in British Columbia, possibly indicating a change in the wintering location of brant.

Great Blue Heron

In Puget Sound, great blue herons (*Ardea herodias*) belong to a non-migratory and marine-oriented subspecies (*A. herodias* subsp. *fannini*) that ranges from Alaska to southern Washington state, with the largest concentration occurring in northwestern Washington and southwest British Columbia (Butler 1997). During the non-breeding season, great blue herons are widely dispersed in Puget Sound, utilizing coastal and lowland areas for foraging and roosting. They are often found as solitary individuals. In contrast, between late winter and summer, herons occur in high densities centered on nesting colonies and associated foraging sites. Herons forage in a variety of habitat types depending on local conditions, tides, and season. Saltwater and freshwater marshes provide year-round foraging opportunities of fish, crustaceans, amphibians and reptiles, though terrestrial habitats also provide small mammals in heron diets (Eissinger 2007).

Marine shoreline and intertidal areas are important to the success of coastal heron colonies. In 2004, WDFW performed an aerial survey to determine foraging habitat, distribution, and concentration areas of great blue herons in Puget Sound (Hayes 2006). Based on this survey it was estimated that 73% of the active heron colonies in Puget Sound are directly associated with marine and estuarine intertidal habitats for foraging activities during the breeding season. In particular, the reproductive success of colonies is dependent on prey associated with eelgrass habitats (Eissinger 2007), such as Drayton Harbor, Port Susan, and Samish, Padilla, and Skagit bays.

Few records of historical trends exist for the great blue heron in Puget Sound. Methods for monitoring heron colonies in British Columbia and Puget Sound have recently been developed, although they are not yet standardized between the two areas. In western Washington, colony status has been assessed approximately every four years by WDFW biologists, and larger colonies in certain locations are monitored by independent investigators or conservation groups. Eissinger (2007) conducted a review of available population data and concluded that since the mid-1990s, the population of northwestern great blue herons has been stable, with the current estimate at 4,700 nesting pairs or 9,400 breeding herons in 2003-2004 (Figure 6). This breeding population represents 121 colonies located on Vancouver Island, the British Columbia mainland, and in the Georgia Strait and Puget Sound basins. Notably, approximately 66% of the total population is concentrated in only 16 colonies, and 35% of the total breeding population belongs to five mega-colonies supporting 200-600 breeding pairs each. In the past decade, the Puget Sound population has seen a substantial transformation from a diffuse distribution of smaller colonies across the landscape to larger colonies in upland marine areas. Reasons for this shift are unknown, but possible causal factors include combinations of increased predation by expanding bald eagle population, human disturbance and encroachment on habitat, degradation and fragmentation of nearshore and coastal habitats by development and land use activities, pollution, changes in prey abundance or distribution, and other systemic changes related to ecosystem decline (Eissinger 2007).

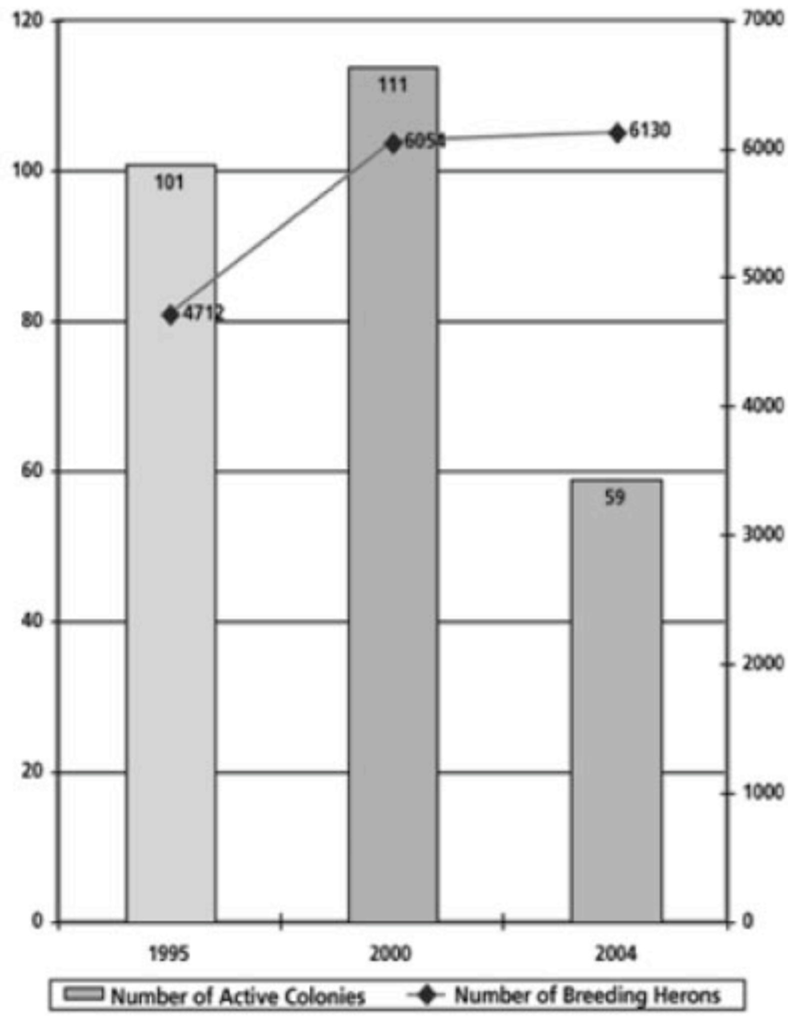


Figure 6. Great blue heron population trends in Puget Sound (reprinted from Eissinger 2007 with permission from the Puget Sound Nearshore Ecosystem Restoration Project and Washington Department of Fish and Wildlife)

Uncertainties

With the recovery of Bald Eagle populations, anecdotal information indicates predation pressure (direct and indirect) has increased at Great Blue Heron colonies. The effect of increasing Bald Eagle presence on colony persistence or productivity by Great Blue Herons is not known.

Trends in waterbird abundance derived from Christmas Bird Counts must be assessed to evaluate whether correction factors that account for observer effort (e.g. party hours) are appropriate. Correction factors applied where they are not necessary could result in a conclusion that abundance had decreased when in fact it had not changed.

Many marine birds migrate, overwinter or breed in regions quire distant from the area(s) they use in Puget Sound. The degree to which potentially significant limiting factors in those areas influence observed changes in abundance in Puget Sound is largely unknown.

Additional work is needed to determine whether changes in abundance of particular marine birds reflect actual population changes or shifts in regional distribution that would locally mimic population declines.

Summary

Multiple species of marine bird that overwinter in Puget Sound have shown sharp declines in abundance over the past two decades. Declining species outnumber increasing species, declines occur across diverse taxonomic groups and feeding guilds, and declines of up to 95% have been reported. Reasons for these declines are not well established and may include factors operating locally, along migration flyways, and at the breeding grounds. Habitat loss and changes in food availability or abundance may have contributed to population changes.

Literature Cited

Anderson, E. M., J. R. Lovvorn, and M. T. Wilson. 2008. Reevaluating Marine Diets of Surf and White-Winged Scoters: Interspecific Differences and the Importance of Soft-Bodied Prey. The Condor 110:285-295.

Bower, J. L. 2009. Changes in marine bird abundance in the Salish Sea: 1975 to 2007. Marine Ornithology 37:9-17.

Butler, R. W. 1997. The great blue heron : a natural history and ecology of a seashore sentinel. UBC Press, Vancouver, BC.

Eissinger, A. M. 2007. Great blue herons in Puget Sound. Puget Sound Nearshore Partnership, Olympia, WA.

Evenson, J. R. 2010. Analysis of marine bird data, Puget Sound Assessment and Monitoring Program. Seattle, WA.

Evenson, J.R, D.R. Nysewander, B.L. Murphie, and T.A. Cyra. 2010. Marine Bird and Mammal Component of the Puget Sound Ambient Monitoring Program web reports. Available at: http://wdfw.wa.gov/mapping/psamp/index.html. Washington Department of Fish and Wildlife, Olympia, WA.

Evenson, J. R., D. R. Nysewander, M. Mahaffy, B. L. Murphie, and T. A. Cyra. 2003. Status, abundance, and colony distribution of breeding pigeon guillemots (*Cepphus columba*) from the inland marine waters of Washington State, as documented by PSAMP efforts, 2000-2002 in Proceedings from Georgia Basin/Puget Sound Research Conference.

Hayes, G. 2006. Puget Sound Great Blue Heron Foraging Aerial Survey 2004. Washington Department of Fish and Wildlife, Olympia, WA.

Jewett, S. G. 1953. Birds of Washington State. University of Washington Press, Seattle.

McShane, C. 2004. Evaluation report for the 5-year status review of the marbled murrelet in Washington, Oregon, and California. EDAW, Inc., Seattle, WA.

Nysewander, D., J. Evenson, B. Murphie, and T. Cyra. 2005. Report of marine bird and mammal component, Puget Sound Ambient Monitoring Program, for July 1992 to December 1999 period. Washington Department of Fish and Wildlife, Olympia, WA.

Ralph, C. J., G. L. Hunt, Jr., M. G. Raphael, and J. F. Piatt. 1995. Ecology and conservation of the marbled murrelet. U.S. Dept. of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, CA.

Raphael, M. G. 2006. Conservation of the Marbled Murrelet under the Northwest Forest Plan. Conservation Biology 20:297-305.

Rathbun, S. F. 1915. List of Water and Shore Birds of the Puget Sound Region in the Vicinity of Seattle. The Auk 32:459-465.

Reed, A., D. H. Ward, D. V. Derksen, and J. S. Sedinger. 1998. Brant (*Branta bernicla*): Birds of North of America.in A. Poole, editor. The Birds of North America Online. Cornell Lab of Ornithology, Ithaca, NY.

Richardson, S. A., D. Hays, R. Spencer, and J. Stofel. 2000. Washington State status report for the common loon. Washington Dept. of Fish and Wildlife, Olympia, WA.

Speich, S. M., T. R. Wahl, and J. F. Watson. 1989. Catalog of Washington seabird colonies. U.S. Dept. of the Interior, Fish and Wildlife Service, Research and Development, Washington, D.C.

Stout, H. A. 2001. Status review of Pacific herring (*Clupea pallasi*) in Puget Sound, Washington. U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, WA.

Vermeer, K. 1981. Food and populations of Surf Scoters in British Columbia. Wildfowl 32:107-116.

Wahl, T. R. 1981. Marine bird populations of the Strait of Juan de Fuca, Strait of Georgia, and adjacent waters in 1978 and 1979. National Oceanic and Atmospheric Administration; United States Environmental Protection Agency, Research and Development, Seattle, WA; Washington, D.C.; Springfield, VA.

Wahl, T. R. 2002. Trends in numbers of marine birds wintering on Bellingham Bay. Washington Birds 8:29-40.

Wilson, U. W. 2005. The Effect of the 1997-1998 El Nino on Rhinoceros Auklets on Protection Island, Washington. Condor 107:462-468.

Wilson, U. W., and D. A. Manuwal. 1986. Breeding Biology of the Rhinoceros Auklet in Washington. Condor 88:143-155.

Bald eagles

Background

Bald eagles (*Haliaeetus leucophalus*) are present year-round throughout most parts of Washington with the highest densities in the Puget Sound region. Individuals occur in the Puget Sound basin as migrants, winter residents and members of the breeding population. They are often associated with shorelines and large, open expanses of water (Stalmaster 1987). Bald eagles are opportunistic foragers that feed most frequently on fish and waterfowl, and as both predators and kleptoparasites, possess a variety of hunting behaviors, consuming live fish and birds as well as scavenging upon dead fish (particularly salmonids), birds and mammals (Watson 2002, Stinson et al. 2007). They are known to hunt in both seabird (Kaiser 1989, Thompson 1989) and great blue heron colonies (Norman et al. 1989).

Breeding bald eagles require large trees near open water in locations that experience relatively low levels of human activity. In Washington, surveys by Washington Department of Fish and Wildlife (WDFW) conducted in 2005 showed that nearly all (97 %) of surveyed bald eagle nests were within 3,000 feet of shoreline (Stinson et al. 2007). While nests are most numerous near marine shorelines, many are also found on shores of lakes, reservoirs, and rivers of Washington. In a more detailed study of 53 breeding pairs throughout western Washington from 1986 - 1997, Watson et al. (2002) found that the mean home range size of 53 bald eagle pairs distributed across lakes, marine shorelines, rivers and bays was 4.9 km2, and ranged from approximately 2 to 7 km2. The density of nesting eagles depends on many factors that affect habitat quality including prey populations, degree of human disturbance, and the availability of nest and perch trees.

Breeding pairs initiate nesting activities in January or February and disperse by late summer when many migrate north to coastal British Columbia and southeast Alaska for several weeks to take advantage of food supplies associated with late summer and early fall salmon runs (Watson 1998). The timing of breeding activities in Washington has been summarized by Watson (2006). Fledglings also disperse northward, but they may remain there for several months before returning to Washington.

Washington's wintering eagles begin to arrive in October from northern breeding territories in Alaska and Canada. Most adults arrive in November and December and many juveniles arrive in January (Buehler 2000, Watson 2001). The winter distribution of bald eagles in Washington is similar to the breeding distribution, but more concentrated at salmon spawning streams and waterfowl wintering areas. Winter ranges are considerably larger and more variable than breeding ranges.

Threats to bald eagles include habitat degradation and reductions in prey such as salmonids in Puget Sound and its surrounding watershed. Alteration of upland nesting habitat from natural events (e.g., windstorms) or human-related factors (e.g., timber harvest, development) that results in either mortality or reduced availability of nest trees or suitable territories, can reduce the number of occupied nesting territories. Because average life expectancy of nests can be shorter than that of breeding birds (Stalmaster 1987), bald eagles often need trees of similar stature located nearby to serve as replacement nest trees if a nesting territory is to persist at the site.

Conservation Status

Bald eagles in Washington were listed as Threatened under the federal Endangered Species Act (ESA) in 1978. The widespread use of DDT between the 1940s and 1970s is widely viewed as the main cause of the decline of bald eagles in Washington and the other 48 states, though direct extirpation and habitat alteration are also known causes (Stalmaster 1987). In response to rebounding populations, the bald eagle was removed from protection under the ESA in 2007 (USFWS 2007a). The bald eagle is still protected by the Bald and Golden Eagle Protection Act and the Migratory Bird Treaty Act (USFWS 2007b). At the state level, bald eagles were down-listed to Sensitive status by the Washington Fish and Wildlife Commission. Habitat protection is still authorized in Washington by the Bald Eagle Protection Law of 1984 (RCW 77.12.655), which requires the establishment and enforcement of rules for buffer zones around bald eagle nest and roost sites. Habitat is protected through bald eagle management plans approved by WDFW. Between1986 and 2005, over 2,900 bald eagle plans were developed between WDFW and various landowner entities for activities on private, state, and municipal lands in Washington (Stinson et al. 2007).

Status

The most recent statewide breeding season census conducted by WDFW, in 2005, found 840 occupied nests in 1,125 territories searched (Stinson et al. 2007). Breeding activity was confirmed by the presence of eggs or shells in or around the nest or observations of adults incubating eggs or brooding chicks.

Trends

WDFW began localized monitoring of bald eagle nests in the San Juan Islands in the early 1960s. The first extensive survey that covered Washington's entire marine shoreline was conducted in 1975 and statewide comprehensive activity and productivity surveys were conducted annually from 1980-1992. Nest activity surveys were continued through 1998, and conducted again in 2001 and 2005. From 1981 to 2005 the nesting population in Washington had increased seven fold (Figure 1)(Stinson et al. 2007). The number of bald eagle territories in Puget Sound also increased substantially (Figure 2)(Stinson et al. 2007). As of 2010, there were 751 known territories in the Puget Sound Basin, with most sites occurring in San Juan, Clallam, Island and Skagit counties (Table 1). Although historical estimates of the bald eagle population are not available, Stinson et al. (2007) estimated 1,328 serviceable breeding locations (SBL; analogous to a territory) existed in Washington prior to European settlement. We note, however, that the estimate of SBLs included various assumptions that cannot be evaluated relative to conditions of the 19th century. The number of known territories in 2010 was 1,403 (WDFW database), which suggests the population may be at or near carrying capacity. While the carrying capacity of bald eagles in Washington is not known, a recent decline in nest occupancy rate suggests that nesting habitat in parts of western Washington may be approaching saturation (Stinson et al. 2007). The number of resident breeders, and trends in localized winter counts suggest that Washington state hosts approximately 4,000 resident and migratory bald eagles each winter (Stinson et al. 2007).

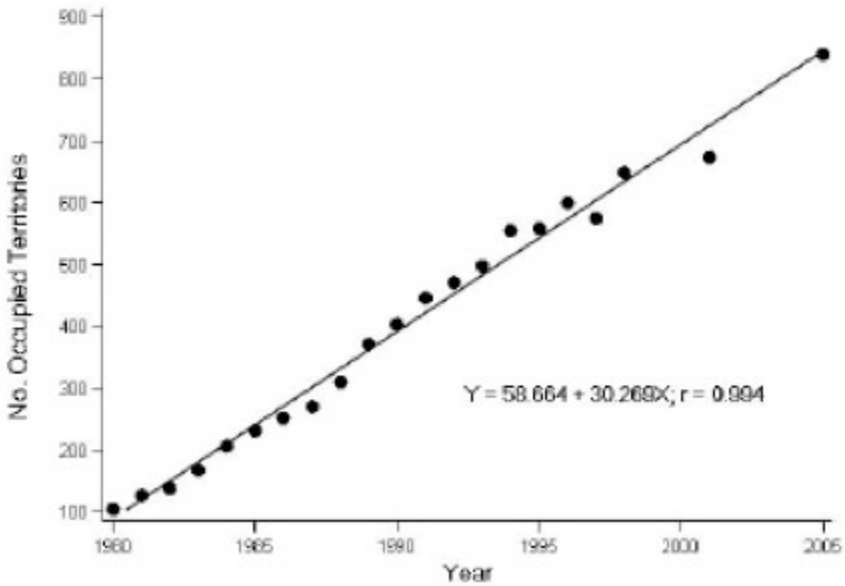


Figure 1. Time trend in population status (number of occupied nests), 1980-2005 (reprinted from Stinson et al. 2007 with permission from Washington Department of Fish and Wildlife)

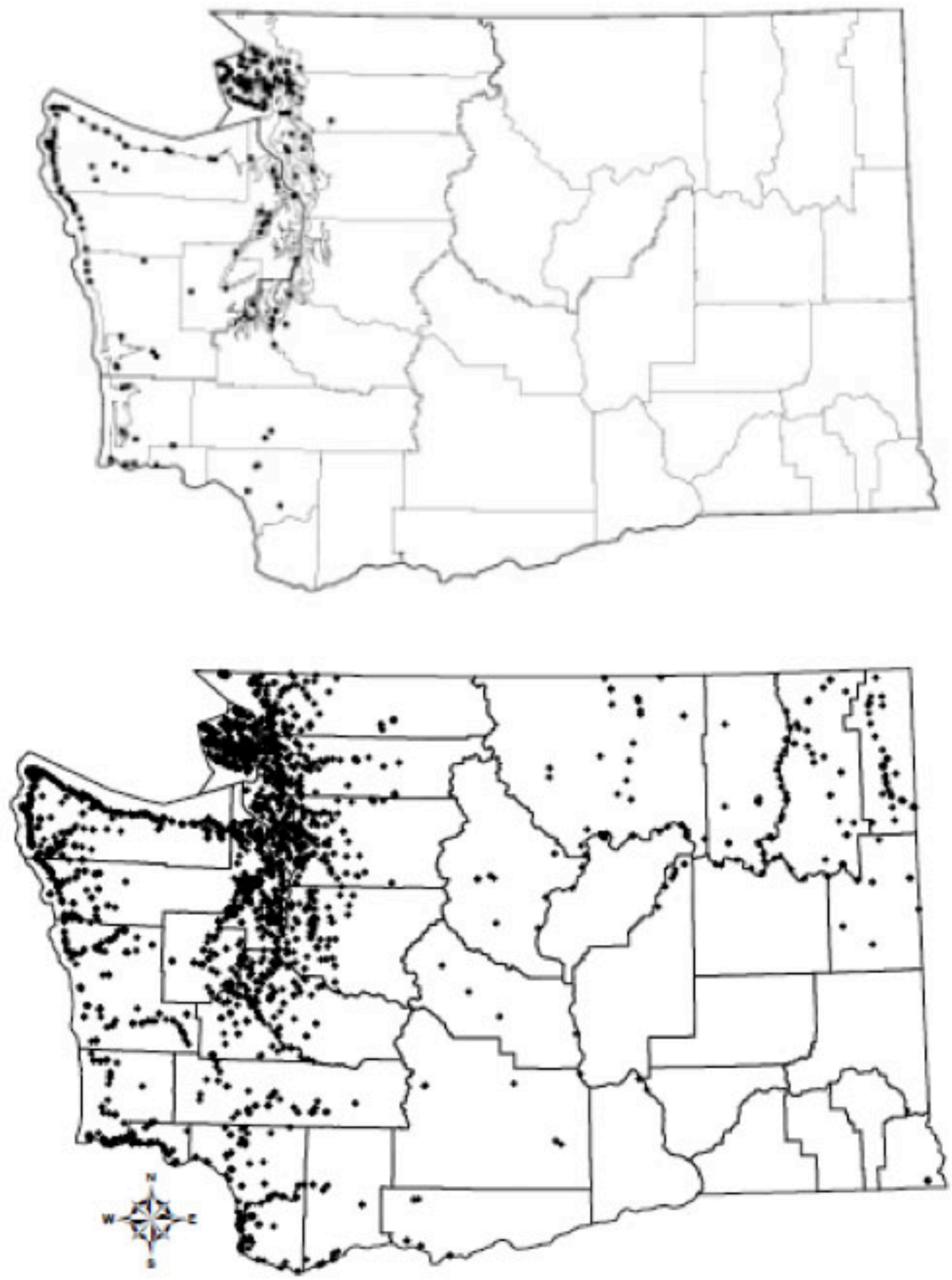


Figure 2. Distribution of bald eagle nests in Washington, in 1980 (top map) and 2005 (bottom map)(reprinted from Stinson et al. 2007 with permission from Washington Department of Fish and Wildlife)

Table 1. Number of bald eagle territories in those portions of Washington counties that are included in the Puget Sound basin. Data from WDFW database, methods according to Stinson et al.2007.

County	Number of Territories
Clallam	85
Island	84
Jefferson	59
King	51
Kitsap	69
Mason	33
Pierce	51
San Juan	98
Skagit	82
Snohomish	57
Thurston	17
Whatcom	65
Total	751

Uncertainties

1. The carrying capacity of bald eagles is unknown and likely varies from one ecosystem type or condition to another. Future monitoring will be necessary to identify carrying capacity.

2. Because bald eagles are closely associated with the marine environment, they are potentially vulnerable to contaminants in the marine food chain. The extent to which they may be vulnerable and the specific contaminant groups that might influence their physical or behavioral health are unknown.

3. The human population is expected to increase substantially in the next three decades. Much of the increase in Washington's population will likely occur in the Puget Basin. Potential responses to increased human pressures on habitats associated with nest territories, and the ability of existing rules to protect those habitats given increasing human pressures, are unknown.

Summary

Bald eagle abundance in Washington has increased in the past three decades, likely in response to federal and state management efforts. The number of nesting pairs in Washington is approximately eight times the number present when the use of DDT was restricted in 1972 (Stinson et al. 2007). In Puget Sound, the predicted rise in human population will continue to increase pressure on nesting and roosting habitats. State bald eagle protection rules (WAC 232-

12-292), along with other forest clearing regulations, may allow the population to persist at or near its current level.

Literature Cited

Buehler, D. A. 2000. Bald eagle (*Haliaeetus leucocephalus*) Birds of North America No. 506. 40 pp. Birds of North America, Philadelphia, PA.

Kaiser, G. W. 1989. Nightly concentration of bald eagles at an auklet colony. Northwestern Naturalist 70:12-13.

Norman, D. M., A. M. Breault, and I. E. Moul. 1989. Bald Eagle incursions and predation at great blue heron colonies. Colonial Waterbirds 12:215-217.

Stalmaster, M. V. 1987. The bald eagle. Universe Books, New York.

Stinson, D. W., J. W. Watson, and K. R. McAllister. 2007. Washington State status report for the bald eagle. Washington Dept. of Fish and Wildlife, Wildlife Program, Olympia, WA.

Thompson, S. P. 1989. Observations of bald eagles eating glaucous-winged gull eggs in western Washington. Northwestern Naturalist 70:13-14.

USFWS. 2007a. Removing the bald eagle in the Lower 48 states from the list of Endangered and Threatened wildlife: Final Rule. Federal Register 72: 37346-37372.

USFWS. 2007b. Authorization under the Bald and Golden Eagle Protection Act for take of eagles. Federal Register 72:31141-31155.

Watson, J. W. 2002. Comparative home ranges and food habits of bald eagles nesting in four aquatic habitats in western Washington. Northwestern Naturalist 83:101-108.

Watson, J.W. 2006. Bald eagle nesting chronology in western Washington. Washington Birds 9:8-11.

Watson, J. W., and D. J. Pierce. 1998. Ecology of bald eagles in western Washington with an emphasis on the effects of human activity. Washington Department of Fish and Wildlife, Olympia.

Watson, J. W., and D. J. Pierce. 2001. Skagit River bald eagles: movements, origins, and breeding population status. Washington Department of Fish and Wildlife, Olympia.

Harbor Seals

BackgrouNd

Harbor seals (*Phoca vitulina*) are found throughout temperate and arctic waters of the northern hemisphere, and inhabit coastal and estuarine waters along the eastern Pacific Ocean from Baja California north to the Gulf of Alaska and Bering Sea (Carretta et al. 2004, Carretta et al. 2007) . Harbor seals are found throughout the nearshore waters of Washington including Hood Canal, Puget Sound, the San Juan Islands, and the Strait of Juan de Fuca out to Cape Flattery (Jeffries et al. 2003) (Figure 1). They use hundreds of locations in Puget Sound to haul out or rest, including intertidal rocks, reefs, and beaches, logbooms, docks and floats. Harbor seals in Washington are considered non-migratory and display strong fidelity to haulout sites. Their local movements are associated with tidal cycles, time of day, weather, and prey availability (Zamon 2001, Carretta et al. 2004, Hayward et al. 2005, Carretta et al. 2007, Patterson and Acevedo-Gutierrez 2008). Most individuals in the inland waters forage in close proximity to haulout sites, and return to the same areas for foraging and haulout (Lance and Jeffries 2006). In general, harbor seals forage opportunistically on prey that are locally and seasonally abundant(Lance and Jeffries 2006, 2007).

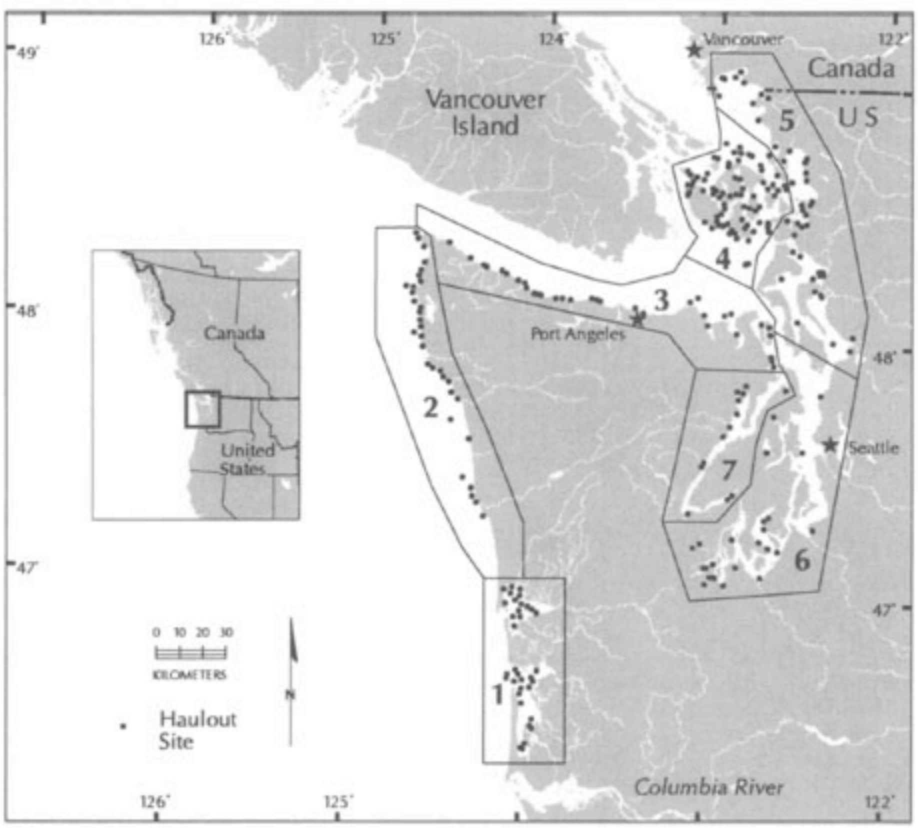


Figure 1. Map of harbor seal haulout sites and survey regions for Washington. The inland stock includes the Strait of Juan de Fuca (3), San Juan Islands (4), Eastern Bays (5), Puget Sound (6), and Hood Canal (7) (reprinted with permission from Jeffries et al. 2003).

Threats to harbor seals include incidental takes from drift gillnet fisheries, vessel strikes, and contaminants. Harbor seals are vulnerable to contamination by persistent organic pollutants (POPs) because they are long-lived, occupy a high trophic level, and have limited metabolic capacity to eliminate pollutants (Ross et al. 2004). Exposure to contaminants has also been associated with immunotoxicity and outbreaks of infectious disease (Mos et al. 2006). Harbor seals in Puget Sound are also heavily contaminated with polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) (Simms et al. 2000, Ross et al. 2004, Cullon et al. 2005).

Status

Harbor seal numbers were severely reduced during the first half of the twentieth century by a state-financed population control program. This bounty program ceased in 1960, and in 1972, harbor seals became protected under the federal Marine Mammal Protection Act (MMPA) and by Washington State. Based on morphological, phenological and genetic differences, the coastal and inland populations of Washing are considered to be two different stocks (Carretta et al. 2007). Currently, both the inland and coastal stocks of harbor seals are not considered "depleted" under the MMPA or listed as "threatened" or "endangered" under the ESA. Population count data collected using aerial surveys of haulouts conducted by WDFW in 1999 indicate both stocks to be within their Optimum Sustainable Population (OSP) ranges as defined by Jeffries et al. (2003).

Trends

It is estimated that 2,000-3,000 harbor seals resided in Washington in the early 1970s (Newby 1973), and historic population levels prior to this are unknown. Beginning in 1983, WDFW initiated consistent aerial surveys of harbor seal inland waters population, which continued through 1999. Jeffries et al.(2003) found that during 1999, Washington inland stock contained 13, 692 seals and that both the coastal and inland populations were near carrying capacity (Figure 2). Thus, at the population levels of 1999, Jeffries et al. (2003) estimated that Washington State harbor seal populations could withstand significant declines and still be within the Optimum Sustainable Population levels. The 1999 population count continues to be the most recent estimate of Washington harbor seal abundances (Carretta et al. 2007).

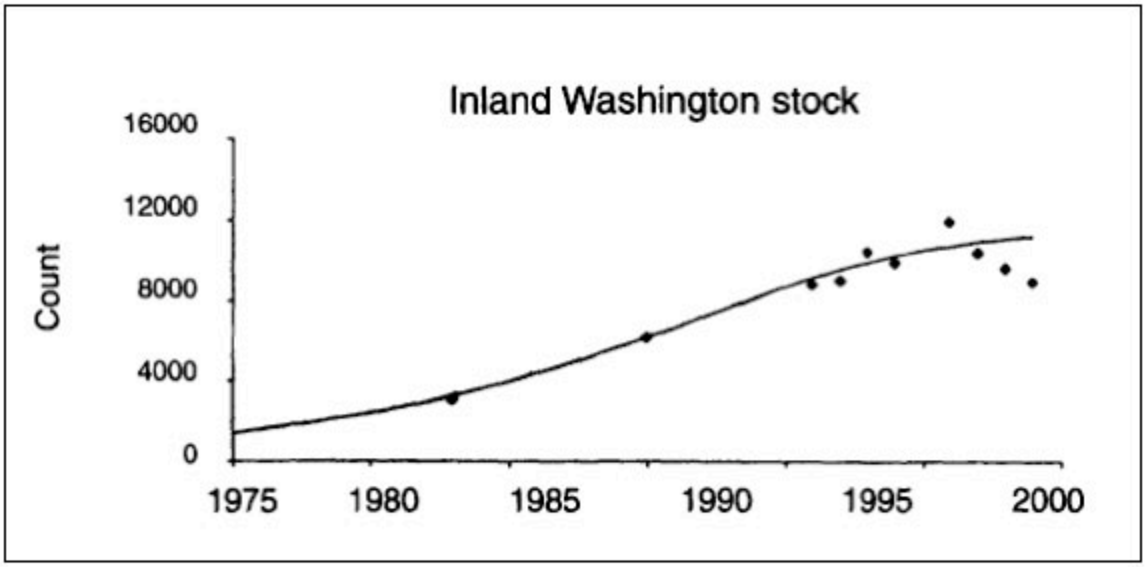


Figure 2. Generalized logistic growth curve of aerial counts of harbor seals in inland waters of Washington (includes the Strait of Juan de Fuca, East Bays, San Juan Islands, Hood Canal, and Puget Sound regions) (reprinted with permission from Jeffries et al. 2003).

Uncertainties

Harbor seal abundance estimates are based on aerial surveys of maximum haul-out counts, which can be complicated by spatial and temporal variability in the behavior of the seals and in the proportion of individuals that are observable (i.e., onshore) during sampling events. To address uncertainty in the proportion onshore, current estimates of trends and population abundances (Jeffries et al. 2003) use both a static correction factor developed by Huber et al. (2001) and an observation-error time series model fitting using maximum likelihood techniques to estimate population dynamic model parameters. To address variability in seal behavior, Hayward et al. (2005) suggest an environmentally dynamic modeling approach, but this has not been adopted. The impacts of contaminant exposure on population status are not well known.

Summary

Harbor seals populations in Washington State have recovered since the 1970s and population sizes may be near a stable equilibrium level, perhaps reflective of the current carrying capacity of the environment. Because of their high trophic position, harbor seal contaminant loads may be used as indicators of <u>pollution</u> levels in Puget Sound (Ross et al. 2004), and have been suggested as possible indicators of other types of anthropogenic change (climate change, fishing activities) (Hindell et al. 2003) and fish community composition (Lance and Jeffries 2007).

Literature Cited

Carretta, J. V., K. A. Forney, M. M. Muto, J. Barlow, J. Baker, B. Hanson, and M. Lowry. 2007. United States marine mammal stock assessments: 2006. National Marine Fisheries Service, La Jolla, CA.

Carretta, J. V., K. A. Forney, M. M. Muto, J. Barlow, J. Baker, and M. Lowry. 2004. U.S. Pacific Marine Mammal Stock Assessments: 2003. NOAA Technical Memorandum NMFS-SWFSC-358.

Cullon, D. L., S. J. Jeffries, and P. S. Ross. 2005. Persistent organic pollutants in the diet of harbor seals (*Phoca vitulina*) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): A food basket approach. Environmental Toxicology and Chemistry 24:2562-2572.

Hayward, J., S. M. Henson, C. J. Logan, C. R. Parris, M. W. Meyer, and B. Dennis. 2005. Predicting numbers of hauled-out harbour seals: a mathematical model. Journal of Applied Ecology 42:108-117.

Hindell, M. A., C. J. A. Bradshaw, R. G. Harcourt, and C. Guinet. 2003. Ecosystem monitoring: Are seals a potential tool for monitoring change in marine systems? Page 446 in N. Gales, M. Hindell, and R. Kirkwood, editors. Marine mammals: fisheries, tourism, and management issues. CSIRO, Collingwood, VIC.

Huber, H. R., S. J. Jeffries, R. F. Brown, R. L. DeLong, and G. VanBlaricom. 2001. Correcting aerial survey counts of harbor seals (*Phoca vitulina richardsi*) in Washington and Oregon. Marine Mammal Science 17:276-293.

Jeffries, S., H. Huber, J. Calambokidis, and J. Laake. 2003. Trends and status of harbor seals in Washington state: 1978-1999. Journal of Wildlife Management 67:207-218.

Lance, M. M., and S. J. Jeffries. 2006. Estimating importance of rockfish, lingcod and other bottomfish in the diet of harbor seals in the San Juan Islands. Page 20. Contract Report to SeaDoc Society, Olympia.

Lance, M. M., and S. J. Jeffries. 2007. Temporal and spatial variability of harbor seal diet in the San Juan Island archipelago. Research Agreement No. K004431-25, Contract report to SEAdoc society research agreement NO. K004431-25. Washington Department of Fish and Wildlife, Olympia, WA.

Mos, L., B. Morsey, S. J. Jeffries, M. B. Yunker, S. Raverty, S. De Guise, and P. S. Ross. 2006. Chemical and biological pollution contribute to the immunological profiles of free-ranging harbor seals. Environmental Toxicology and Chemistry 25:3110-3117.

Newby, T. C. 1973. Changes in Washington State harbor seal population, 1942-1972. Murrelet 54:5-6.

Patterson, J., and A. Acevedo-Gutierrez. 2008. Tidal influence on the haul-out behavior of harbor seals (*Phoca vitulina*) at a site available at all tide levels. Northwestern Naturalist 89:17-23.

Ross, P. S., S. J. Jeffries, M. B. Yunker, R. F. Addison, M. G. Ikonomou, and J. C. Calambokidis. 2004. Harbor seals (*Phoca vitulina*) in British Columbia, Canada, and Washington State, USA, reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry 23:157-165.

Simms, W., S. Jeffries, M. Ikonomou, and P. S. Ross. 2000. Contaminant-related disruption of vitamin A dynamics in free-ranging harbor seal (*Phoca vitulina*) pups from British Columbia, Canada, and Washington State, USA. Environmental Toxicology and Chemistry 19:2844-2849.

Zamon, J. E. 2001. Seal predation on salmon and forage fish schools as a function of tidal currents in the San Juan Islands, Washington, USA. Fisheries Oceanography 10:353-366.

Killer Whales

Background

Three distinct groups of killer whales (*Orcinus orca*) occupy the coastal waters of the northeastern Pacific. These groups—northern and southern residents, transients, and offshores—are distinguished by diet, behavior, morphology, and other characteristics. Among these, Southern Resident and transient killer whales commonly are found in Puget Sound. Northern residents and offshore killer whales rarely enter Puget Sound (Wiles 2004, Kriete 2007), and therefore are not described in detail here.

While the taxonomic status of north Pacific killer whales remains unresolved (summarized in Krahn et al. 2004, NMFS 2008), the Southern Resident killer whale (SRKW) and transient killer whale populations are considered by NOAA to be separate stocks based on genetic, morphological, dietary and behavioral differences and are classified as endangered (SRKW) and threatened (transient) under the U.S. Endangered Species Act (2005). The SRKW population is found primarily in Washington and southern British Columbia and includes three groups or pods (J-, K- and L-pod) (Krahn et al. 2002, Krahn et al. 2004). Their home range during the spring, summer, and fall includes Puget Sound, the Strait of Juan de Fuca, and the Strait of Georgia (NMFS 2008) (Figure 1). During the late fall to winter, SRKWs travel as far south as central California and north to the Queen Charlotte Islands, British Columbia. The distribution of transient killer whales ranges from southern California to Icy Strait and Glacier Bay in Alaska (Ford et al. 2000). Transients are recorded along the Puget Sound and Vancouver Island shorelines during the summer and early fall (Wiles 2004).

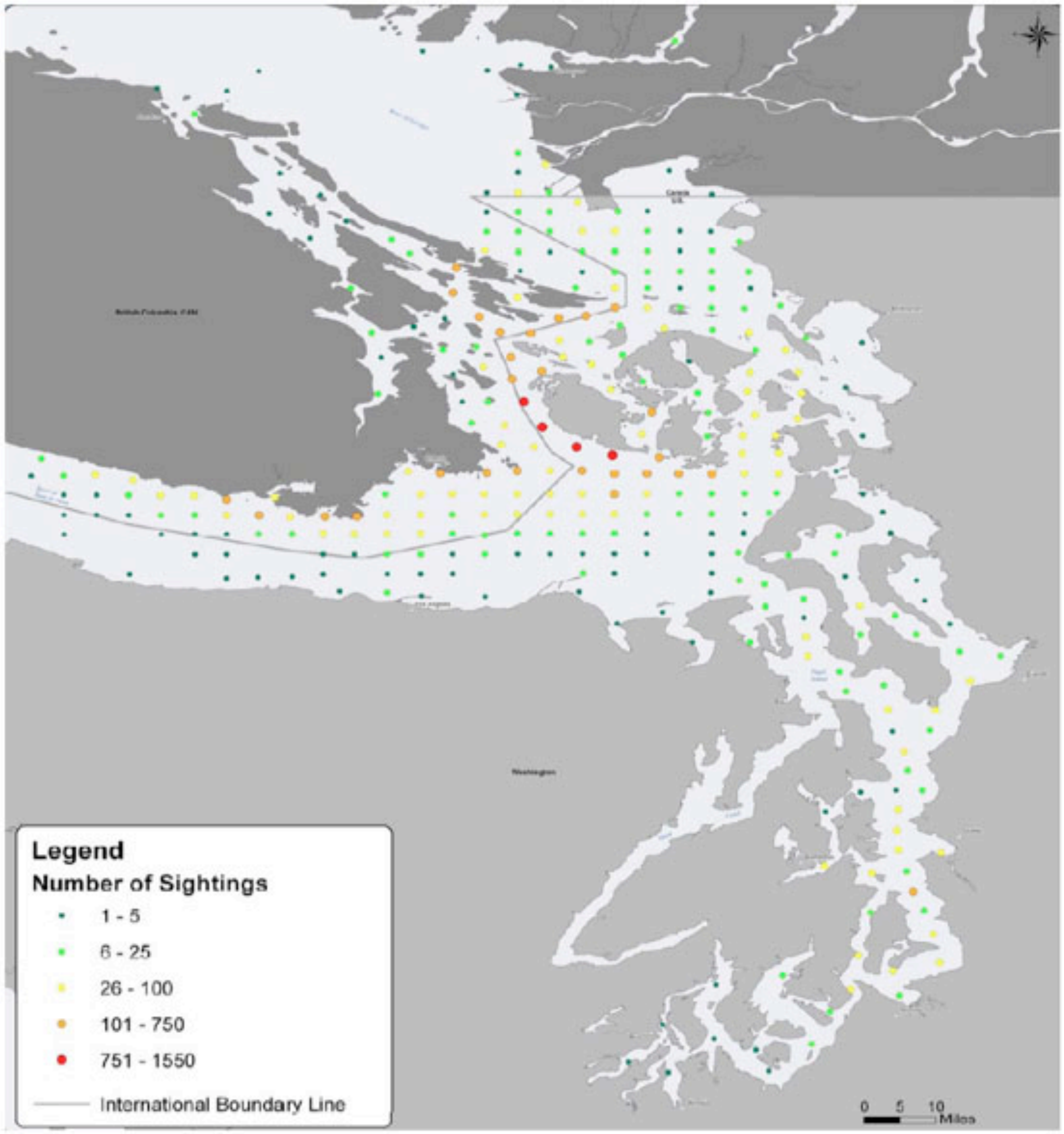


Figure 1. Distribution of Southern Resident killer whale sightings from 1990-2005 (data from The Whale Museum 2005; figure reprinted from NMFS 2008, courtesy of NOAA Fisheries).

Resident killer whales are believed to principally consume marine fish, while transients prey solely on marine mammals (Ford et al. 1998, Ford et al. 2000). Diets of resident killer whales were found to include 22 species of fish and one species of squid (Ford et al. 1998). A detailed dietary study based on 529 observed predation events from 1997 – 2005 of both Northern and Southern Resident killer whales revealed that salmonids (particularly Chinook) comprised 96% of the killer whale diet. However, most of these observations (>85%) were based on Northern residents; less information is available on the Southern Residents that routinely inhabit Puget Sound (Ford and Ellis 2006). The diet of transient killer whales is less well known, but is thought to be comprised primarily of harbor seals and to include other marine mammals such as sea lions, harbor porpoise, Dall's porpoise, minke whales and marine birds (Ford et al. 1998).

The movements and locations of SRKWs have been recorded by researchers, whale watchers and citizens since the early 1970s and a database of their distribution is maintained by The Whale Museum in Friday Harbor, Washington. Whales are most frequently observed in the San Juan Archipelago but are also found as far into Puget Sound as the southern portion of the South Sound (Figure 1)(Hauser et al. 2007, NMFS 2008). Southern Resident pods are present regularly in the Georgia Basin, and during warmer months all pods concentrate their activity from the south side of the San Juan Archipelago through Haro Strait northward to Boundary Pass (Hauser et al. 2007). Most transient sightings in the Puget Sound-Georgia Basin region are concentrated around southeastern Vancouver Island, the San Juan Archipelago, and the southern edge of the Gulf Islands. Transients appear to utilize a wider range of water depths and habitats than residents (NMFS).

Three main factors have been identified as potential threats to killer whales in Washington and British Columbia: reductions in prey availability, disturbance by underwater noise and vessel traffic, and exposure to environmental contaminants, particularly PCBs and PBDEs (NMFS 2008). Ford et al. (2010) suggests that declines in SRKW abundance in the mid 1990s were driven by a significant decline in range-wide abundance of Chinook salmon. NMFS has published a Final Recovery Plan that describes a recovery program designed to address each of the threats to the SRKW population. Due to their trophic position as apex predator, levels of contaminants such as polychlorinated biphenyl (PCBs) and dioxins in both resident and transient killer whales have been shown to be among the highest recorded (Ross et al. 2000, Krahn et al. 2007, Krahn et al. 2009).

Status

SRKW population: Photo-identification censuses of the SRKW population performed by the Center for Whale Research since the 1970s have shown several periods of growth and decline (Figure 2). Because the average life expectancy of killer whales is estimated to be 50 years and can extend to 80-90 years, the existing data on the SRKW populations have covered only a small portion of the lifespan. In response to a 20% population decline from 1996 to 2001, the SRKW stock was designated as depleted under the Marine Mammal Protection Act (MMPA) in 2003 and became listed as Endangered under the Endangered Species Act (ESA) in 2005. In 2006, NMFS designated approximately 2,500 square miles as critical habitat for Southern Residents. The designated area encompasses parts of Haro Strait, the waters around the San Juan Archipelago, the Strait of Juan de Fuca, and all of Puget Sound.

Transient population: Detailed estimates of population abundances for transient killer whale populations have not been made (NMFS 2008). It is hypothesized that historical transient killer whale populations experienced a large decline in abundance due to substantial prey losses in the early-to-mid 1900s (Springer et al. 2003). Because harbor seal populations in the region have increased over the last 30 years and currently are close to carrying capacity (Jeffries et al. 2003), it is believed that transients are no longer prey-limited (Ford et al. 2000). Approximately 225 transients have been identified in Washington, British Columbia, and southeastern Alaska (NMFS 2008) although current abundances are not known (NMFS 2008).

Trends

SRKW population: The historical population of Southern Residents in the mid- to late-1800s ws estimated to be approximately 200 whales (Krahn et al. 2002), although lack of data prior to the 1970s makes contributes to the uncertainty of this estimate. The capture of live killer whales for aquaria is thought to have removed approximately 50 Southern Resident and 5 transient killer whales between 1962 and 1977 (NMFS 2008). Since that time, the population has experienced fluctuations with periods of positive population growth followed by decline (Figure 2). Most notable was a substantial period of population growth between the mid 1980s and mid 1990s, during which total whale numbers expanded from 75 to nearly 98 animals. That period was followed by a brief period of decline, to 80 animals, followed by a moderate increase thereafter (Wiles 2004, Kriete 2007, NMFS 2008). The most recent estimate of 85 animals derives from a survey conducted in April 2009 (Center for Whale Research, (reported in PSP 2009)(Figure 2).

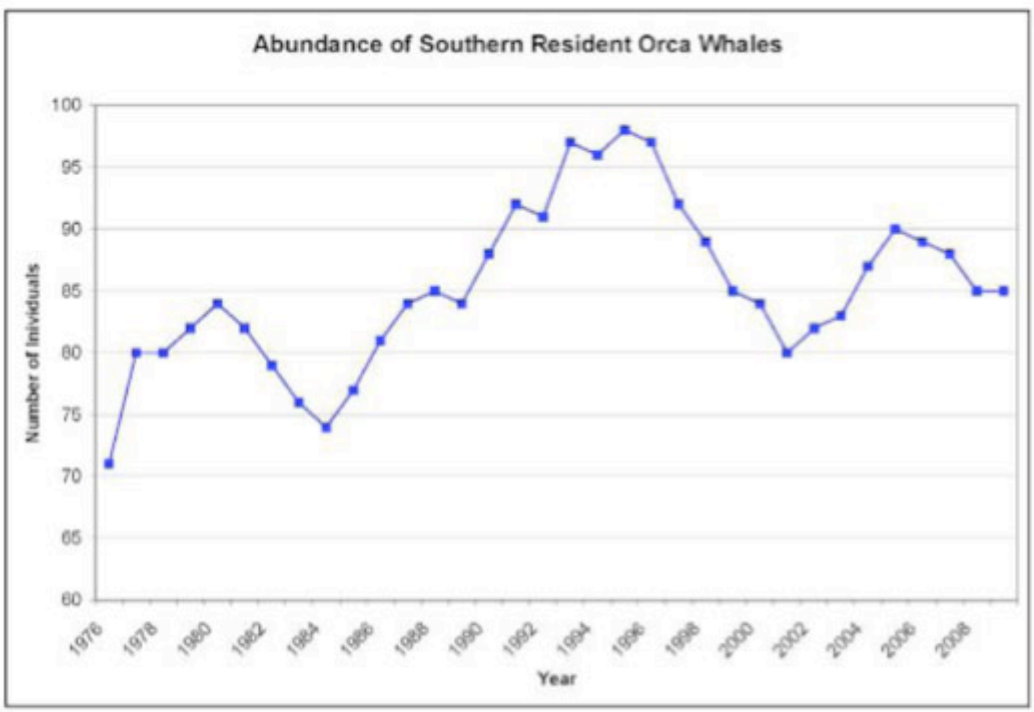


Figure 2. Abundance of Southern Resident killer whales from 1976-2009 (data from the Center for Whale Research)(reprinted from PSP 2009)

SRKW population predictions: Krahn et al. (2004) conducted a population viability analysis (PVA)(Morris and Doak 2002) to assess the future risk of extinction of the SRKW population, the predictions of which varied significantly according to the time period from which survival rates were estimated. Using the survival rates estimated from 1974-2003, they found that extinction probabilities for the SRKW whale populations ranged from <0.1-3% over the next 100 years and 2-42% over the next 300 years. However, extinction probabilities based on 1994-2003 survival rates ranged from 6-19% over the next 100 years and 68-94 % over the next 300 years (Krahn et al. 2004)

Transient population: Trends in abundance of the transient killer whale population cannot be estimated because accurate assessments of transient killer whale abundance have not been made.

Uncertainties

While the diets of Northern resident killer whales, which inhabit the coastal habitat of British Columbia and Alaska, have been well characterized (Ford and Ellis 2006), the extent to which diets of Northern resident killer whales are predictors of the diets of SRKW population (the primary users of Puget Sound habitats) remains under investigation. There is strong evidence for correlations between fluctuations in salmonids, especially Chinook salmon, and resident killer whales (Ford and Ellis 2006), but the drivers behind this relationship have not been elucidated. Furthermore, the unknown and potentially interactive effects of multiple stressors on killer whales introduces uncertainty in projections of future population abundances.

Summary

Killer whales are challenging to study because they spend much of their time below the water surface, are wide-ranging, and are highly migratory. Photo-identification and vigilant observations of predation events have allowed researchers to identify every individual in the SRKW population based on unique patterns and morphology, thereby facilitating accurate estimation of population abundance and diet of Resident killer whales. Human removal of SRKW appears to have driven population declines prior to the 1970s, yet 35 years after the removals for live capture ended, SRKW population numbers remain low. Data on transient killer whale populations are lacking.

Literature Cited

Ford, J. K. B., and G. M. Ellis. 2006. Selective foraging by fish-eating killer whales *Orcinus orca* in British Columbia. Marine Ecology-Progress Series 316:185-199.

Ford, J. K. B., G. M. Ellis, and K. C. Balcomb. 2000. Killer whales: The natural history and genealogy of *Orcinus orca* in British Columbia and Washington. UBC Press; University of Washington Press, Vancouver; Seattle.

Ford, J. K. B., G. M. Ellis, L. G. Barrett-Lennard, A. B. Morton, R. S. Palm, and K. C. Balcomb. 1998. Dietary specialization in two sympatric populations of killer whales (*Orcinus orca*) in coastal British Columbia and adjacent waters. Canadian Journal of Zoology-Revue Canadienne De Zoologie 76:1456-1471.

Ford, J. K. B., G. M. Ellis, P. F. Olesiuk, and K. C. Balcomb. 2010. Linking killer whale survival and prey abundance: food limitation in the oceans' apex predator? Biology Letters 6:139-142.

Hauser, D. D. W., M. G. Logsdon, E. E. Holmes, G. R. VanBlaricom, and R. W. Osborne. 2007. Summer distribution patterns of southern resident killer whales *Orcinus orca*: core areas and spatial segregation of social groups. Marine Ecology-Progress Series 351:301-310.

Jeffries, S., H. Huber, J. Calambokidis, and J. Laake. 2003. Trends and status of harbor seals in Washington state: 1978-1999. Journal of Wildlife Management 67:207-218.

Krahn, M. M., M. J. Ford, W. F. Perrin, P. R. Wade, R. P. Angliss, M. B. Hanson, B. L. Taylor, G. M. Ylitalo, M. E. Dahlheim, J. E. Stein, and R. S. Waples. 2004. 2004 Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. Page 73 in N. T. Memorandum, editor.

Krahn, M. M., M. B. Hanson, R. W. Baird, D. G. Burrows, C. K. Emmons, J. K. B. Ford, L. L. Jones, D. P. Noren, P. S. Ross, G. S. Schorr, T. K. Collier, and R. H. Boyer. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54:1903-1911.

Krahn, M. M., M. B. Hanson, G. S. Schorr, C. K. Emmons, D. G. Burrows, J. L. Bolton, R. W. Baird, and G. M. Ylitalo. 2009. Effects of age, sex and reproductive status on persistent organic pollutant concentrations in "Southern Resident" killer whales. Marine Pollution Bulletin 58:1522-1529.

Krahn, M. M., P. R. Wade, S. T. Kalinowski, M. E. Dahlheim, B. L. Taylor, M. B. Hanson, G. M. Ylitalo, R. P. Angliss, J. E. Stein, and R. S. Waples. 2002. Status review of Southern Resident killer whales (*Orcinus orca*) under the Endangered Species Act. Page 133 in N. T. Memorandum, editor.

Kriete, B. 2007. Orcas in Puget Sound. Puget Sound Nearshore Partnership, Seattle.

Morris, W. F., and D. F. Doak. 2002. Quantitative conservation biology : theory and practice of population viability analysis. Sinauer Associates, Sunderland, Mass.

NMFS. 2008. Recovery plan for Southern Resident killer whales (*Orcinus orca*). National Marine Fisheries Service, Seattle.

PSP. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB Concentrations in Free-Ranging Pacific Killer Whales, *Orcinus orca*: Effects of Age, sex and dietary preference. Marine Pollution Bulletin 40:504-515.

Springer, A. M., J. A. Estes, G. B. van Vliet, T. M. Williams, D. F. Doak, E. M. Danner, K. A. Forney, and B. Pfister. 2003. Sequential megafaunal collapse in the North Pacific Ocean: An ongoing legacy of industrial whaling? Proceedings of the National Academy of Sciences of the United States of America 100:12223-12228.

Wiles, G. J. 2004. Washington State status report for the killer whale. Washington Department of Fish and Wildllife, Olympia, WA.

HABITATS

1. Eelgrass

Background

Eelgrass (*Zostera marina* L.) is an aquatic flowering plant common in tidelands and shallow waters along much of Puget Sound's shoreline. The species is restricted to soft-sediment habitats. Sexual reproduction occurs through seed production. Vegetative spread occurs via growth of below-ground rhizomes, which can result in the formation of large, dense beds. Eelgrass is widely recognized for its provision of important ecological functions (e.g., Hemminga and Duarte 2000, Duarte 2002), which in Puget Sound include the provision of energy to sustain diverse nearshore food webs (e.g., Simenstad and Wissmar 1985), as well as the creation of structurally complex habitat for a suite of species including herring, crab, shrimp, shellfish, waterfowl, and salmonids (Simenstad 1994, Heck et al. 2003, Mumford 2007). Eelgrass also stabilizes sediments and minimizes shoreline erosion (Duarte 2002). Because eelgrass requires growing conditions that include good water clarity and low nutrients, eelgrass abundance is considered to be an important indicator of estuarine health (e.g., Dennison et al. 1993, Hemminga and Duarte 2000). Industrial, agricultural and residential practices in upland areas and watersheds, and particularly activities that increase inputs of nutrients and suspended sediments, can negatively impact the growth of eelgrass. Direct physical impacts to eelgrass, such as propeller scour, overwater structures and shoreline development, also pose threats (Mumford 2007). In Washington, *Z. marina* has been designated a species of special concern by WDFW (WAC 220-110-250) and as critical habitat by the WDOE Shoreline Management Act (RCW 90.58).

Characteristics in Puget Sound

Eelgrass occurs in shallow soft sediments habitats throughout much of Puget Sound, with the notable exception of the southernmost portion (Mumford 2007, Gaeckle et al. 2009). Two habitat types are distinguishable based on nearshore geomorphology. Eelgrass flats are expansive, shallow beds typically located in bays, but also found at river deltas and shoals. Eelgrass fringe habitats consist of comparatively narrow, linear beds that follow the shoreline. In Puget Sound, eelgrass fringe habitats are more common than eelgrass flats, but because some flats are large in areal extent, the total area occupied by eelgrass is distributed roughly equally between the two habitat types. In the north Puget Sound and Saratoga-Whidbey regions, eelgrass occurs predominantly in large flats in Padilla and Samish Bays, which together account for approximately 25% of the total eelgrass in Puget Sound. By contrast, in the central, southern, and Strait of Juan de Fuca regions of Puget Sound, fringe beds are more common.

Multiple factors determine eelgrass distribution, including substrate availability, water clarity, wave energy, light attenuation, water temperature, tidal amplitude, and desiccation stress (Hemminga and Duarte 2000). In Puget Sound, the maximum depth to which eelgrass grows ranges from approximately 1.3 m below the low tide line (MLLW) to greater than 9 m deep. The deepest beds are found in the Strait of Juan de Fuca and the San Juan Islands (Gaeckle et al. 2009) .

WDNR Eelgrass Monitoring

The Nearshore Habitat Program of the Washington State Department of Natural Resources (WDNR) monitors eelgrass distribution and abundance through the Submerged Vegetation Monitoring Project (SVMP). The SVMP was established in 2000 to better understand eelgrass resources throughout Puget Sound and to detect temporal changes the distribution and abundance of eelgrass. The SVMP is part of the Puget Sound Assessment and Monitoring Program (PSAMP), a multi-agency effort coordinated by the Puget Sound Partnership to monitor diverse physical and biotic aspects of the Puget Sound ecosystem. Eelgrass is sampled annually at approximately 100 randomly-selected sites and 6 six permanent “core” sites (Figure 1). Sampling is performed at three spatial scales: Sound-wide, within regions, and within individual sites. Since monitoring began in 2000, more than 270 sites have been assessed. The SVMP was designed to detect changes that occur at annual and longer-term (5- and 10-year) temporal scales. The SVMP’s primary programmatic performance measure is the ability to detect a 20% decline in eelgrass abundance over 10 years at the Sound-wide scale (Berry et al. 2003, Gaeckle et al. 2009). Data collection is carried out using underwater videography recorded along transects. Twelve to fifteen transects are sampled per site, oriented perpendicular to shore and sampled using a line-intercept method.

Figure 1. Distribution of site sampled in 2008 by SVMP sound -wide eelgrass monitoring study (reprinted from Gaeckle et al.2009 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

Status

Currently about 22,800 ± 4,500 hectares of eelgrass exist in greater Puget Sound, occupying approximately 43% of Puget Sound shoreline (Gaeckle et al. 2009). Eelgrass is more abundant in north Puget Sound than in the south. Approximately 91% of the estimated 9,859 ±2,603 hectares occurs in large, shallow embayments (Gaeckle et al. 2009). At individual sites, the areal extent of eelgrass ranges from less than 1 hectare to more than 3,000 hectares.

Trends

Trends in eelgrass distribution and abundance in Puget Sound prior to 2000 are difficult to establish due to a lack of long-term and broad-scale information preceding the initiation of the SVMP. Thom and Hallum (1990) performed a comprehensive examination of historical hydrographic charts, aerial photographs, WDFW survey information and other limited observations of eelgrass distribution in Puget Sound. The authors reported apparent declines in eelgrass abundance since the late 1800s in Bellingham Bay and the Snohomish River Delta, and an apparent increase in eelgrass abundance over approximately the same period in Padilla Bay.

Since monitoring began in 2000, the SVMP reports that the total area occupied by eelgrass in the Puget Sound has remained relatively stable (Gaeckle et al. 2009)(Figure 2). Despite this, site-level analyses suggest that in seven out of the last eight sampling periods, declines have been more frequent than increases (Gaeckle et al. 2009, Puget Sound Partnership 2009)(Table 1 and Figure 3), and sites with long-term declines outnumber sites with long-term increases (Gaeckle et al. 2009). Declines generally have occurred at smaller sites, while the extensive beds in the region, such as Padilla Bay and Samish Bay, remained stable. Gaeckle et al. (2009) conclude that the SVMP data suggests an overall pattern of slight declines in eelgrass throughout Puget Sound.

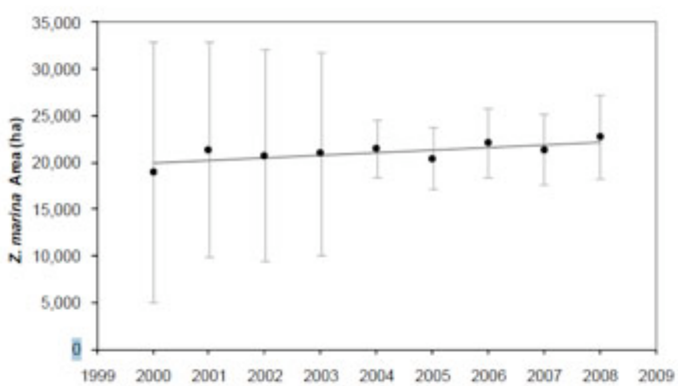


Figure 2. Sound-wide changes in area occupied by eelgrass from 2000 to 2008. Error bars represent 95% confidence intervals. The sharp improvement in precision in 2004 is due to increased sampling frequency at large sites. (reprinted from Gaeckle et al. 2009 with permission from Nearshore Habitat Program, Washington Department of Natural Resources)

Table 1. Results of a multiple parameter assessment of regional Z. marina condition based on data collected from 2000-2008. The number of measurable changes within a region was quantified and compared to the number of significant positive or negative changes (alpha = 0.05). CPS = Central Puget Sound, HDC = Hood Canal, NPS = North Puget Sound, SJS = San Juan/Straits, SWH = Saratoga/Whidbey. From Gaeckle et al. 2009, Nearshore Habitat Program, Washington Department of Natural Resources.

	CPS				HDC				NPS				SJS				SWH			
	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change	No. Change Tests	Significant change	Positive change	Negative change
Site-level area	147	10	1	9	72	15	2	13	70	8	3	5	133	14	1	13	89	10	4	6
Deep edge depth	122	12	4	8	69	8	0	8	65	7	4	3	112	12	2	10	83	8	5	3
Shallow edge depth	122	16	7	9	68	13	5	8	64	8	4	4	111	12	6	6	83	15	7	8
5-year area trends	25	6	2	4	15	6	0	6	16	2	1	1	19	6	2	4	13	2	0	2
Proportion of significant results	0.11				0.19				0.12				0.12				0.14			
Proportion of significant positive results	0.32				0.16				0.48				0.25				0.43			
Proportion of significant negative results	**0.68**				**0.84**				**0.52**				**0.75**				**0.57**			

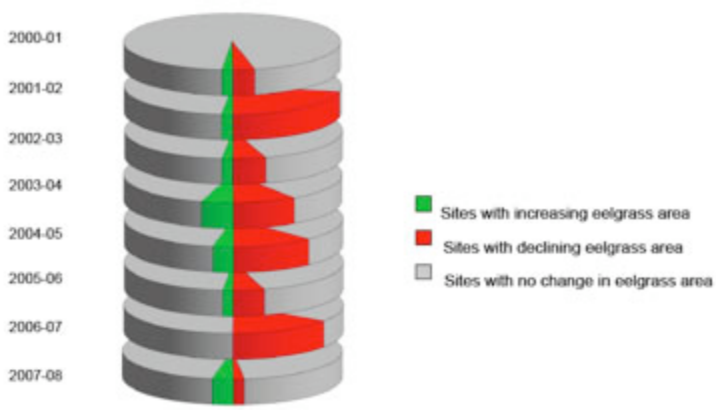


Figure 3. Eelgrass changes at individual sites. In seven of eight years of annual change, a greater proportion of sites showed statistically significant declines compared with increases in eelgrass area.(Nearshore Habitat Program, Washington Department of Natural Resources; reprinted from PSP 2009)

Uncertainties

The relative importance of the factors driving fluctuations in the distribution and abundance of eelgrass in Puget Sound is not well understood. Changes in key abiotic factors such as water clarity and nutrient levels may be important, yet analyses linking such abiotic data to eelgrass abundances have not been conducted. Consequently, the causes for declines in eelgrass cover documented by the SVMP are not known, nor are the ecological consequences of such declines for the taxa that utilize eelgrass habitat such as birds, invertebrates and fishes.

Summary

Eelgrass is critically important for maintaining nearshore ecosystem function and is recognized as a valuable indicator of ecosystem health. While the overall aerial extent of eelgrass in Puget Sound has shown no significant change over the past eight years, sharp local declines have been reported at some sites. The causes of these declines have not been established.

Literature Cited

Berry, H. D., A. T. Sewell, S. Wyllie-Echeverria, B. R. Reeves, T. F. Mumford, J. R. Skalski, R. C. Zimmerman, and J. Archer. 2003. Puget Sound Submerged Vegetation Monitoring Project: 2000-2002 Monitoring Report, Nearshore Habitat Program. Washington State Department of Resources, Olympia, WA.

Dennison, W. C., R. J. Orth, K. A. Moore, J. C. Stevenson, V. Carter, S. Kollar, P. W. Bergstrom, and R. A. Batiuk. 1993. Assessing water quality with submersed aquatic vegetation. BioScience 43:86-94.

Duarte, C. M. 2002. The future of seagrass meadows. Environmental Conservation 29:192-206.

Gaeckle, J. L., P.Dowty, H. Berry, and L. Ferrier. 2009. Puget Sound Submerged Vegetation Monitoring Project: 2008 Monitoring Report, Nearshore Habitat Program. Washington State Department of Natural Resources, Olympia, WA.

Heck, K. L., Jr., G. Hays, and R. J. Orth. 2003. Critical evaluation of the nursery role hypothesis for seagrass meadows. Marine Ecology Progress Series 253:123-136.

Hemminga, M. A., and C. M. Duarte. 2000. Seagrass Ecology. Cambridge University Press, Cambridge, UK; New York, NY.

Mumford, T. F. 2007. Kelp and eelgrass in Puget Sound. Seattle, WA.

Puget Sound Partnership. 2009. Ecosystem Status & Trends. Puget Sound Partnership.

Simenstad, C. 1994. Faunal associations and ecological interactions in seagrass communities of the Pacific Northwest coast. United States Environmental Protection Agency, Seattle, WA.

Simenstad, C. A., and R. C. Wissmar. 1985. Delta carbon-13 evidence of the origins and fates of organic carbon in estuarine and nearshore food webs. Marine Ecology Progress Series 22:141-152.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests in Puget Sound, Wetland Ecosystem Team. United States Environmental Protection Agency

Kelp

Background

Kelps are large seaweeds in the order Laminariales that form dense canopies in temperate rocky intertidal and subtidal habitats less than 30 m in depth. The kelp flora of the Pacific Northwest is one of the most diverse in the world (Druehl 1969). Kelps are characterized by a highly dimorphic lifecycle consisting of a large diploid sporophytic (bed-forming) phase and a microscopic haploid gametophytic phase. In the Puget Sound region, bull kelp (*Nereocystis luetkeana*) occurs throughout Puget Sound and the Strait of Juan de Fuca, while the distribution of giant kelp (*Macrocystis integrifolia*) is restricted to the Strait of Fuca (Berry et al. 2005, Mumford 2007). Both form conspicuous floating canopies, or kelp beds. Sporophytes of *Nereocystis* are annual or semi-annual, whereas sporophytes of *Macrocystis* are perennial, persisting for several years. In addition to these dominant bed-forming taxa, numerous species of understory (non-floating) kelp occur subtidal habitats, many of which are present in southern and central Puget Sound (Mumford 2007).

Kelps are important primary producers. They contribute to Puget Sound food webs by providing food for herbivores and detritivores, and by releasing dissolved organic carbon (Duggins et al. 1989). In addition, kelps create important biogenic habitat that is utilized by fish, invertebrates, marine mammals, and birds (e.g., Ojeda and Santelices 1984, Graham 2004). Kelp can significantly alter the physical environment by modifying current and wave energy (Eckman et al. 1989) and this buffering capacity can influence the ecology of other organisms that utilize kelp environments for larval dispersal and settlement, for example rockfish (Carr 1991).

The extent and composition of kelp beds varies through time in response to natural and human-induced influences. In general, the distribution of kelp is determined by the amount of light available for photosynthesis, nutrient levels, grazers, physical disturbances, and toxic contaminants (reviewed in Mumford 2007). In addition to these external factors, demographic structure may play an important role in driving temporal dynamics of *Macrocystis* kelp beds through decreased fitness of older, more inbred populations (Raimondi et al. 2004, Reed et al. 2006).

Sea otters have been shown to be keystone predators in kelp forest ecosystem through their consumption of sea urchins, a major grazer of kelps (Estes and Palmisano 1974). In Washington state, otter populations have been slowly increasing since their reintroduction in 1969 and 1970 (Lance et al. 2004) following their extirpation through hunting in the 1900s. While they are more abundant on the open coast, otters have been observed as far east as Pillar Point in the Strait of Juan de Fuca (Lance et al. 2004, Laidre and Jameson 2006) where they have been shown to consumes a high proportion of urchins (Laidre and Jameson 2006). The potential for sea otters to expand into further into Puget Sound could affect kelp populations through trophic interactions. Furthermore, harvest of urchins by humans may be an important indirect driver of kelp populations in the Strait of Juan de Fuca; Berry et al. (2005) anecdotally observed that historic increases in urchin harvest rates were positively associated with increases in kelp abundances. However, in an experimental study, neither simulated fisheries removals nor simulated otter predation significantly affected the abundance of kelps in the San Juan Archipelago (Carter et al. 2007).

In addition to trophic interactions, climate changes associated with El Nino are known to cause short-term declines in kelp populations (e.g., Dayton and Tegner 1984), while the Pacific Decadal Oscillation could be driving changes over longer time periods. Substrate movement, as a result of altered nearshore hydrology and geomorphology, may also influence the amount of available habitat for attachment of kelps (Mumford 2007).

Due to their proximity to shore, kelps are likely to be subjected to anthropogenic impacts such as pollution discharge, nutrient influxes from urban and agricultural sources, increased turbidity, and increased rates of sedimentation (Dayton 1985, Mumford 2007). These can alter photosynthetic performance and growth of sporophytes and prevent settlement, growth, and reproduction of microscopic gametophytes. Toxic contaminants such as petroleum products are known to damage kelp by lowering photosynthetic and respiratory rates in meristematic tissue (Antrim et al. 1995).

Status

The Washington Department of Natural Resources (WDNR) conducts an annual inventory of canopy-forming kelp beds along the outer coast of Washington and the Strait of Juan de Fuca (approximately 360 km of shoreline). Inventories have been conducted annually since 1989 (with the exception of 1993) using aerial color-infrared photography (Van Wagenen 2004). In 2005, Berry et al. (2005) reported a total of approximately 1,700 hectares of floating kelp (Nereocystis and Macrocystis) on Washington's outer coast and the Strait of Juan de Fuca.

Trends

Prior to the initiation of annual floating kelp inventories by WDNR, Thom and Hallum (1990) reviewed several sources of historical data and found evidence that floating kelp had increased by 58 percent since the first European mapping in the 1850s. The largest increases were observed in the most populated areas such as central and south Puget Sound, but anecdotal evidence for losses in central Puget Sound were also noted. Between 1989 and 2004, the annual inventories conducted by WDNR for floating kelp at 66 shoreline sections on the outer coast and in the Strait of Juan de Fuca show high year-to-year variation, ranging from 722 hectares in 1997 to 2,575 hectares in 2000 (Berry et al. 2005)(Figure 1). Between the two species of floating kelp, M. integrifolia canopy area was more stable over time than N. luetkeana canopy, potentially due to their differing life histories. From 1989 to 2004, total floating kelp canopy area increased significantly ($p<0.01$), but these increases were restricted to the Outer Coast and the Western Strait of Juan de Fuca; kelps in the Eastern Juan de Fuca region showed no trend (Berry et al. 2005)(Figure 2). At the smallest scale (5-15km of shoreline), kelp area increased significantly in 18 sections, decreased significantly in 1 section, and did not change significantly in 47 sections (Berry et al. 2005)(Figure 3). A significant decrease in kelp canopy area was detected near Protection Island. Kelp canopies in this area have declined gradually from more than 10 hectares in 1989 and 1990 to less than 1 hectare annually since 1994 (Berry et al. 2005)(Figure 3).

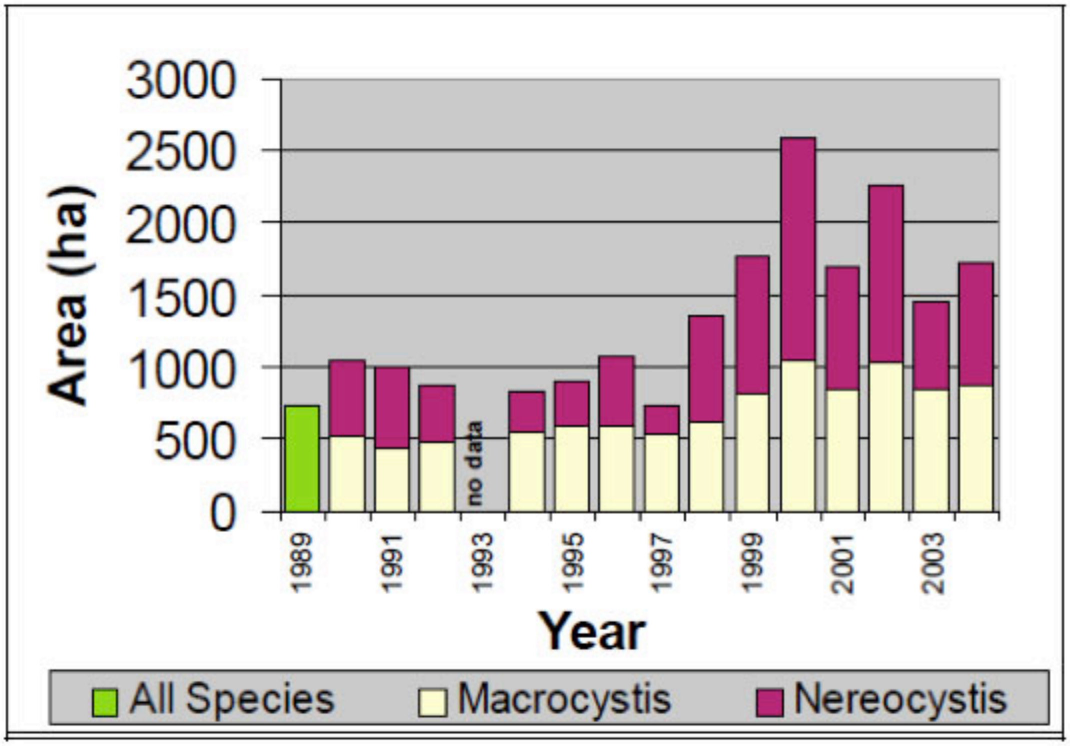


Figure 1. Floating Kelp Canopy Area on Washington's outer coast and the Strait of Juan de Fuca, 1989-2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

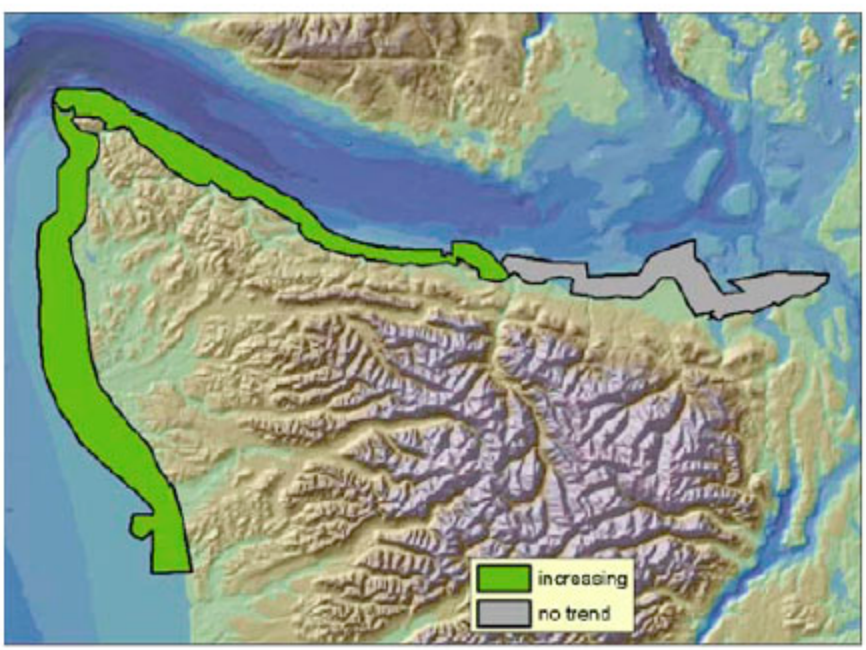


Figure 2. Region changes in kelp canopy area ($p < 0.01$), based on annual surveys between 1989 and 2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

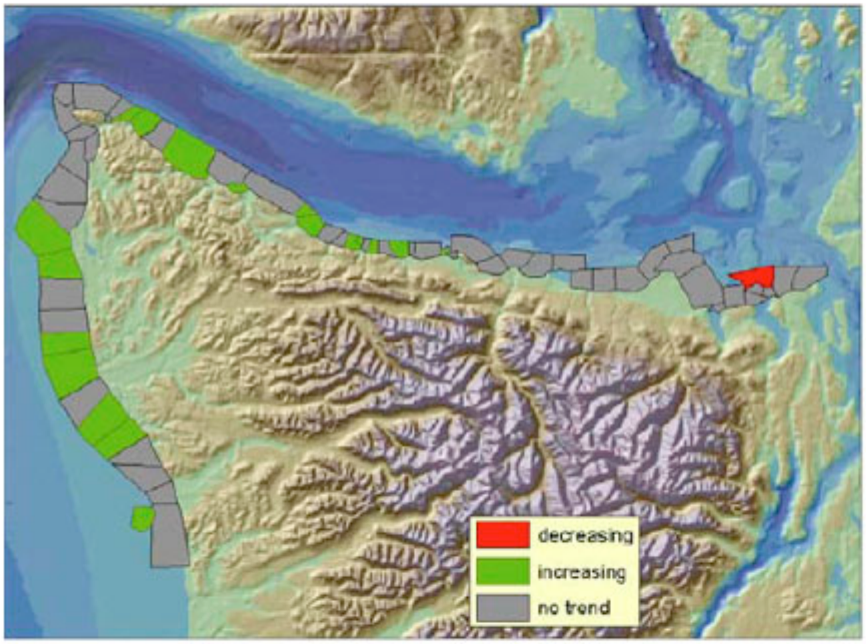


Figure 3. Shoreline sections with significant changes in kelp canopy area ($p < 0.01$), based on annual surveys between 1989 and 2004 (reprinted from Berry et al. 2005 with permission from Nearshore Habitat Program, Washington Department of Natural Resources).

Despite these findings, Mumford (2007) notes multiple anecdotal accounts of kelp bed losses around Marrowstone, Bainbridge, and Fox islands as well as personal observations of the loss of small kelp beds in southern Puget Sound at Itsami Ledge, Devils Head and Dickenson Point. A large Nereocystis bed on Dallas Bank, north of Protection Island in the Strait of Juan de Fuca, has almost totally disappeared since 1989 (Mumford 2007). Other anecdotal observations indicate substantial declines in bull kelp abundance in the San Juan Archipelago and the Strait of Georgia. Taken together, the observations could suggest widespread declines in bull kelp in Puget Sound. The causes of these changes are not known.

Uncertainties

The long-term WDNR dataset provides important insight into how the aerial extent of kelp canopies has changed over time, yet there may be potential biases associated with this method. Berry et al. (2005) notes that observed trends could be subject to methodological artifacts related to environmental factors (primarily tidal height and current speed) that introduce uncertainty or bias in the monitoring data. Both tides and currents have been shown to affect apparent *Nereocystis* canopy area as observed by photographs taken from the adjacent shoreline (Britton-Simmons et al. 2008). Consequently, it is possible that some of the observed variation in kelp canopy cover may be inflated by changes in the conditions under which the photographs were taken.

The WDNR monitoring programs focuses on the two species of floating kelp (*Nereocystis* and *Macrocystis*) native to the region. However, understory (non-floating) kelps are abundant and widely distributed throughout Puget Sound, where their ecological importance could equal that of the canopy-forming kelps. Effective monitoring of subtidal kelp populations is not yet possible, although use of towed video arrays holds promise (Mumford 2007). Furthermore, little is known about the ecology of the microscopic gametophyte phase of kelps due to the difficulty of studying them in situ (Mumford 2007). Failure in settlement, growth, or reproduction in microscopic stages will result in disappearance of the conspicuous sporophytic phases.

Summary

Kelps are important primary producers and create important biogenic habitat in Puget Sound ecosystems. Annual aerial surveys of floating kelp canopies conducted by WDNR show that between 1898 and 2004 floating canopies increased in outer coastal areas an in the western Strait of Juan de Fuca. Floating kelp canopies in the eastern Strait of Juan de Fuca showed no statistical change over the same period. Anecdotal evidence indicates sharp local declines in kelp abundance in southern and central Puget Sound and the San Juan Archipelago and calls for new investigations and expansion of kelp surveys.

Literature Cited

Antrim, L. D., R. D. Thom, W. W. Gardiner, V. W. Cullinan, D. K. Shreffler, and R. W. Bienert. 1995. Effects of petroleum products on bull kelp (*Nereocystis luetkeana*). Marine Biology 122:23-31.

Berry, H. D., T. F. Mumford, and P. Dowty. 2005. Using historical data to estimate changes in floating kelp (*Nereocystis luetkeana* and *Macrocystis integrifolia*) in Puget Sound, Washington. Puget Sound Georgia Basin Research Conference. Nearshore Habitat Program, Washington Department of Natural Resources, Olympia, WA.

Britton-Simmons, K., J. E. Eckman, and D. O. Duggins. 2008. Effect of tidal currents and tidal stage on estimates of bed size in the kelp *Nereocystis luetkeana*. Marine Ecology Progress Series. 355:95.

Carr, M. H. 1991. Habitat selection and recruitment of an assemblage of temperate zone reef fishes. Journal of Experimental Marine Biology and Ecology 146:113-137.

Carter, S. K., G. R. VanBlaricom, and B. L. Allen. 2007. Testing the generality of the trophic cascade paradigm for sea otters: a case study with kelp forests in northern Washington, USA. Hydrobiologia 579:233-249.

Dayton, P. K. 1985. Ecology of Kelp Communities. Annual Review of Ecology and Systematics 16:215-245.

Dayton, P. K., and M. J. Tegner. 1984. Catastrophic storms, El Nino, and patch stability in a southern California kelp community. Science 224:283-285.

Druehl, L. D. 1969. The northeast Pacific rim distribution of Laminariales. Pages 161-170 in International Seaweed Symposium.

Duggins, D. O., C. A. Simenstad, and J. A. Estes. 1989. Magnification of secondary production by kelp detritus in coastal marine ecosystems. Science 245:170-173.

Eckman, J. E., D. O. Duggins, and A. T. Sewell. 1989. Ecology of under story kelp environments. I. Effects of kelps on flow and particle transport near the bottom. Journal of Experimental Marine Biology and Ecology 129:173-187.

Estes, J. A., and J. F. Palmisano. 1974. Sea Otters: Their role in structuring nearshore communities. Science 185:1058-1060.

Graham, M. H. 2004. Effects of Local Deforestation on the Diversity and Structure of Southern California Giant Kelp Forest Food Webs. Ecosystems 7:341-357.

Laidre, K. L., and R. J. Jameson. 2006. Foraging Patterns and Prey Selection in an Increasing and Expanding Sea Otter Population. Journal of Mammalogy 87:799-807.

Lance, M. M., S. A. Richardson, and H. L. Allen. 2004. Washington State Recovery Plan for the Sea Otter. Washington Dept. of Fish and Wildlife, Wildlife Program, Olympia, WA.

Mumford, T. F. 2007. Kelp and eelgrass in Puget Sound. Seattle District, U.S. Army Corps of Engineers, Puget Sound Nearshore Partnership, Seattle, WA.

Ojeda, F. P., and B. Santelices. 1984. Invertebrate communities in holdfasts of kelp *Macrocystis pyrifera* from Southern Chile. Marine Ecology Progress Series 16:65-73.

Raimondi, P. T., D. C. Reed, B. Gaylord, and L. Washburn. 2004. Effects of self-fertilization in the giant kelp, *Macrocystis pyrifera*. Ecology 85:3267-3276.

Reed, D. C., P. T. Raimondi, L. Washburn, B. Gaylord, B. P. Kinlan, and P. T. Drake. 2006. A metapopulation perspective on patch dynamics and connectivity in giant kelp.in J. P. Kritzer and P. F. Sale, editors. Marine Metapopulations. Elsevier Academic Press, Amsterdam; Boston.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests in Puget Sound, Wetland Ecosystem Team. United States Environmental Protection Agency.

Van Wagenen, R. 2004. Washington coastal kelp resources, Summer 2004. Nearshore Habitat Program, Washington Department of Natural Resources, Olympia, WA.

Tidal Wetlands

Background

Tidal wetlands are highly productive ecosystems that provide a variety of resources and ecosystem functions to Puget Sound biota and humans. In this report, tidal wetlands refer to both estuarine (intertidal) and riverine tidal (tidally-influenced freshwater) wetlands along the Puget Sound shoreline. Wetlands provide important ecosystem roles, directly regulating hydrologic and biogeochemical processes and supporing high rates of biological productivity (Mitsch and Gosselink 2007). They also are a key habitat for a suite of fish, amphibian, invertebrate and bird species including chum and Chinook salmon, herring, Dungeness crabs and Great Blue Herons (e.g., McMillan et al. 1995, Simenstad and Cordell 2000, Eissinger 2007 , Stick and Lindquist 2009). Because of the fjord-like topography in Puget Sound, tidal wetlands are predominantly associated with the major rivers. The steep, rocky bathymetry and topography limit the existence of extensive intertidal areas or the deposition of sediments on which vegetated wetland might occur (Boule 1981). Low gradient rivers combined with substantial tidal ranges create large intertidal areas in river floodplains that contain plant communities strongly controlled by a substantial amount of freshwater runoff. Tidal wetlands in Puget Sound have experienced significant losses and degradation as a result of development and other land uses.

Status

Collins and Sheikh (2005) characterized tidal wetland habitat across the sub-basins of Puget Sound (Figure 1) using both aerial and oblique photographs taken from 1998 – 2000 as part of a detailed study comparing the extent and nature of current and historical wetlands. They found that nearly half of the current tidal marsh area is in located in the Skagit, Stillaguamish and Samish river deltas and that the median size of a tidal wetland complex is 0.57 hectares (Figure 2)(Collins and Sheikh 2005). They estimate that there are currently 5,650 hectares of tidal wetland habitat in Puget Sound (Collins and Sheikh 2005). In a more detailed analysis of the composition of wetlands in river deltas, they found that the dominant type of tidal wetland in the river deltas of Puget Sound is currently estuarine-emergent wetland relative to the less frequent estuarine scrub-shrub and riverine habitat types (Figure 3)(Collins and Sheikh 2005).

Figure 1. Sub-basins of Puget Sound as defined by Collins and Sheikh (2005). Reprinted with permission from Collins and Sheikh (2005).

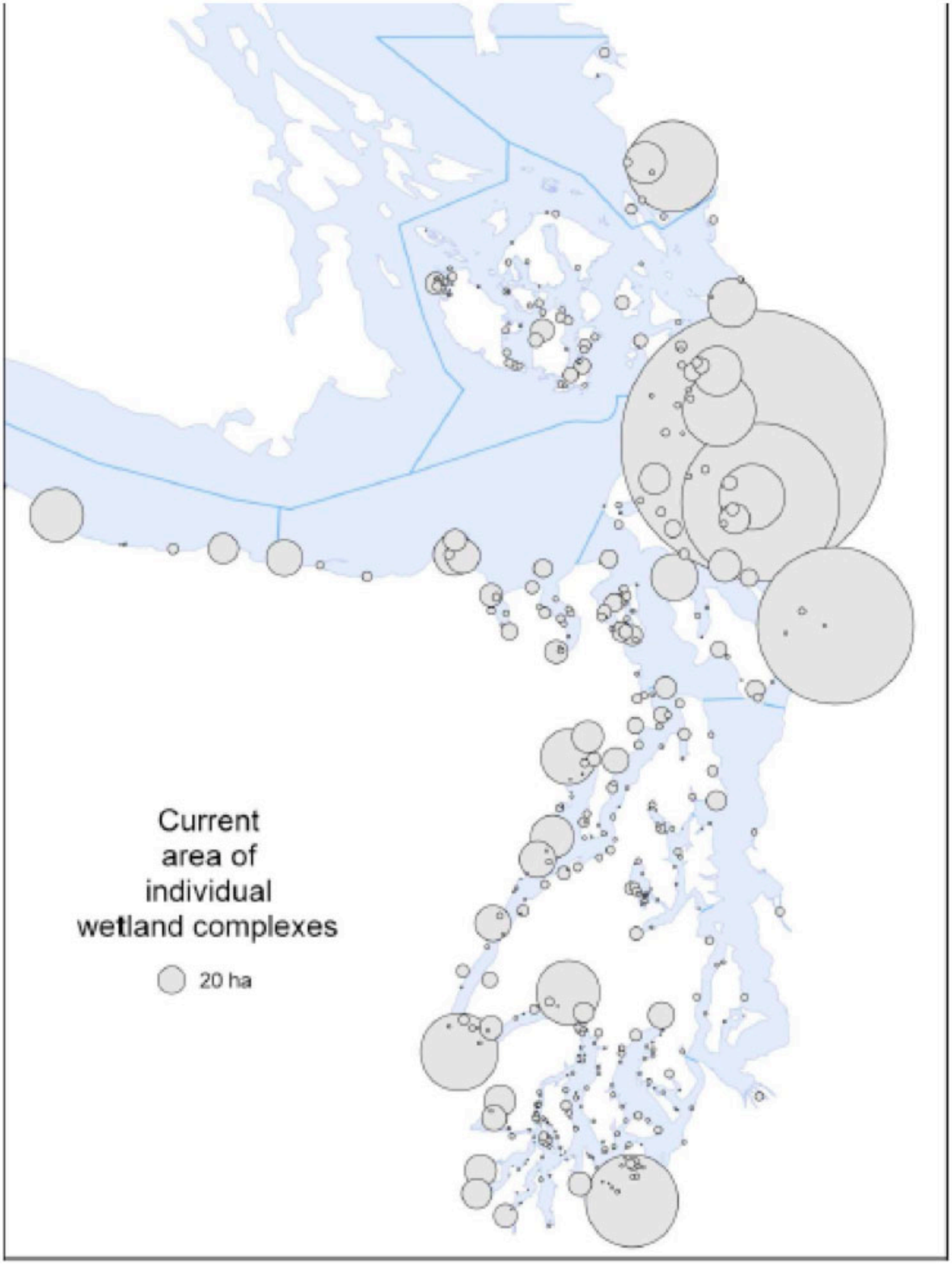

Current
area of
individual
wetland complexes
20 ha

Figure 2. Current area of individual wetland complexes (note: in all pie diagrams, wetland is proportional to the symbol area (reprinted with permission from Collins and Sheikh 2005)

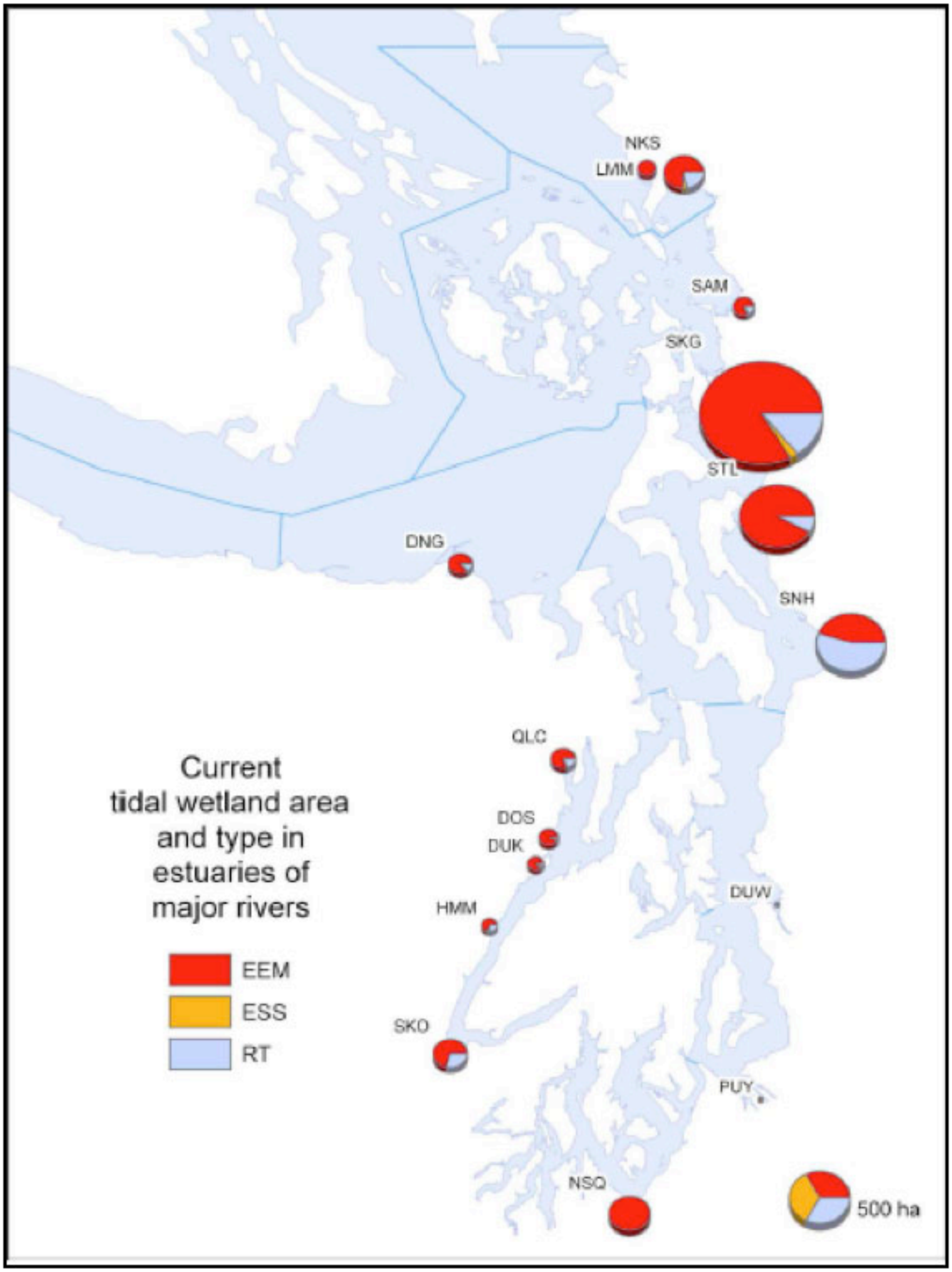


Figure 3. Relative area of current tidal wetland types in the estuaries of major rivers draining the Cascade Range and Olympic Mountains. EEM- estuarine emergent wetland; ESS- estuarine scrub-shrub wetland; RT- riverine-tidal wetland. (note: in all pie diagrams, wetland is proportional to the symbol area) (reprinted with permission from Collins and Sheikh 2005)

Trends

Several quantitative investigations into the degree of alteration of tidal wetlands have been conducted in Puget Sound. The earliest and most comprehensive assessment of areal coverage of tidal wetlands occurred in the mid 1880s by a Snohomish resident for the purposes of assessing agricultural development potential (Nesbit 1885). This endeavor used navigation maps, interviews with residents, and field observations to document the extent of tidal marshes and swamps (inclusive of saltmarsh and freshwater marsh) throughout Washington State from ca.1883. It found that tidal marshes greatly exceeded tidal flats in area on Puget Sound and that freshwater marshes were three to four times as great in extent as compared to the tidal marshes. Based on this early surveying effort by Nesbit (1885) , Thom and Hallum (1990) estimated approximately 26,792 hectares of tidal wetlands in seven of the nine counties bordering Puget Sound in the late 1800s. As such, approximately 38% of tidal marshes in Puget Sound may have already been converted to agricultural and urban land uses by the late 1800s (Nesbit 1885, Collins and Sheikh 2005).

The historic extent of tidal wetlands in Puget Sound was also recorded on topographic charts known as “T-sheets,” which were produced by the U.S. Coast Survey and the U.S. Coast and Geodetic Survey in 1884-1908. A review of comparisons between the T-sheets and more current sources including U.S. Geologic Survey topographic maps (produced in the 1970s) was conducted by Thom and Hallum (1990). This effort also drew upon analyses by Bortelson et al. (1980) and Boule et al. (1983). This investigation revealed that the most substantial intertidal wetland losses occurred in the Snohomish, Duwamish and Puyallup river deltas, reported to have experienced loss of 32 %, 100%, and 99% respectively. Subaerial wetland loss (defined as those wetlands landward of the general saltwater shoreline, but exclusive of intertidal wetlands) was also significant, with total losses of approximately 73% in river deltas throughout Puget Sound since the late 1800s (Bortleson et al. 1980, Thom and Hallum 1990).

More recently, Collins et al. (2003) reconstructed historical environments of several estuaries in northern Puget Sound and concluded that a considerable amount of tidal wetland had already been converted to agricultural and other land uses prior to development of the T-sheets, particularly estuarine scrub-shrub and riverine tidal environments, which were the basis of previous studies. To provide a comprehensive assessment, the Washington Department of Natural Resources (WDNR) collaborated with the University of Washington (UW) to characterize the historic and current distribution, type, and amount of tidal wetlands in Puget Sound (2005). Collins and Sheikh (2005) used a number of other sources that supplemented and cross-referenced the T-sheets, including records of federal land survey, aerial photographs, the survey conducted by Nesbit (1885) and soil surveys. They developed an atlas of pre-settlement (mid 1880s) riverine and nearshore habitats consisting of a spatially explicit digital database based on a landform and process-based classification of nearshore wetlands (see Collins and Sheikh (2005) for a complete summary of methods and results). They estimated the historic area of wetland habitat in Puget Sound to be 29,500 acres, indicating that the current tidal wetlands are 17 – 19% of their historical extent (Collins and Sheikh 2005). They found that the Whidbey basin (which includes the Snohomish, Skagit and Stillaguamish rivers) has experienced the largest total loss of areal coverage followed by the Sand Juan Islands/North Coast (which includes the Padilla Bay part of the greater Skagit River delta, and the Samish River), the Fraser Lowland (which includes the Lummi and Nooksack rivers), and the Central Sound (which includes the Duwamish and Puyallup rivers) (Figure 4). Moreover, the median size of individual wetlands has decreased over time from approximately 0.93 hectares to 0.57 hectares (Figures 2 and 5)(Collins and Sheikh 2005). The composition of river delta wetlands has also undergone a major shift such that the relative abundance of emergent scrub-shrub and riverine-tidal vegetation were historically higher than current levels (Figures 3 and 6)(Collins and Sheikh 2005).

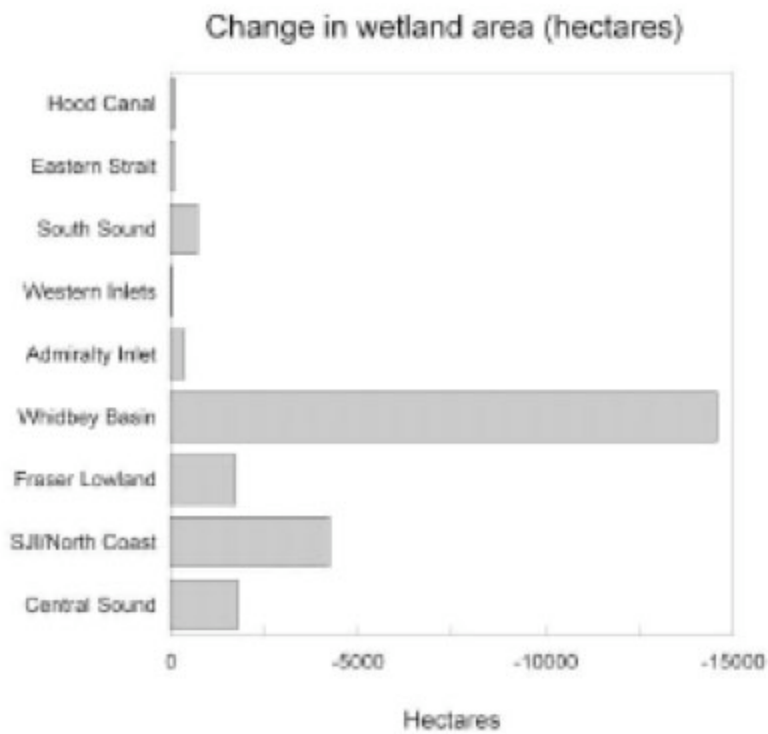


Figure 4. Change in wetland area (hectares) in Puget Sound (reprinted with permission from Collins and Sheikh 2005)

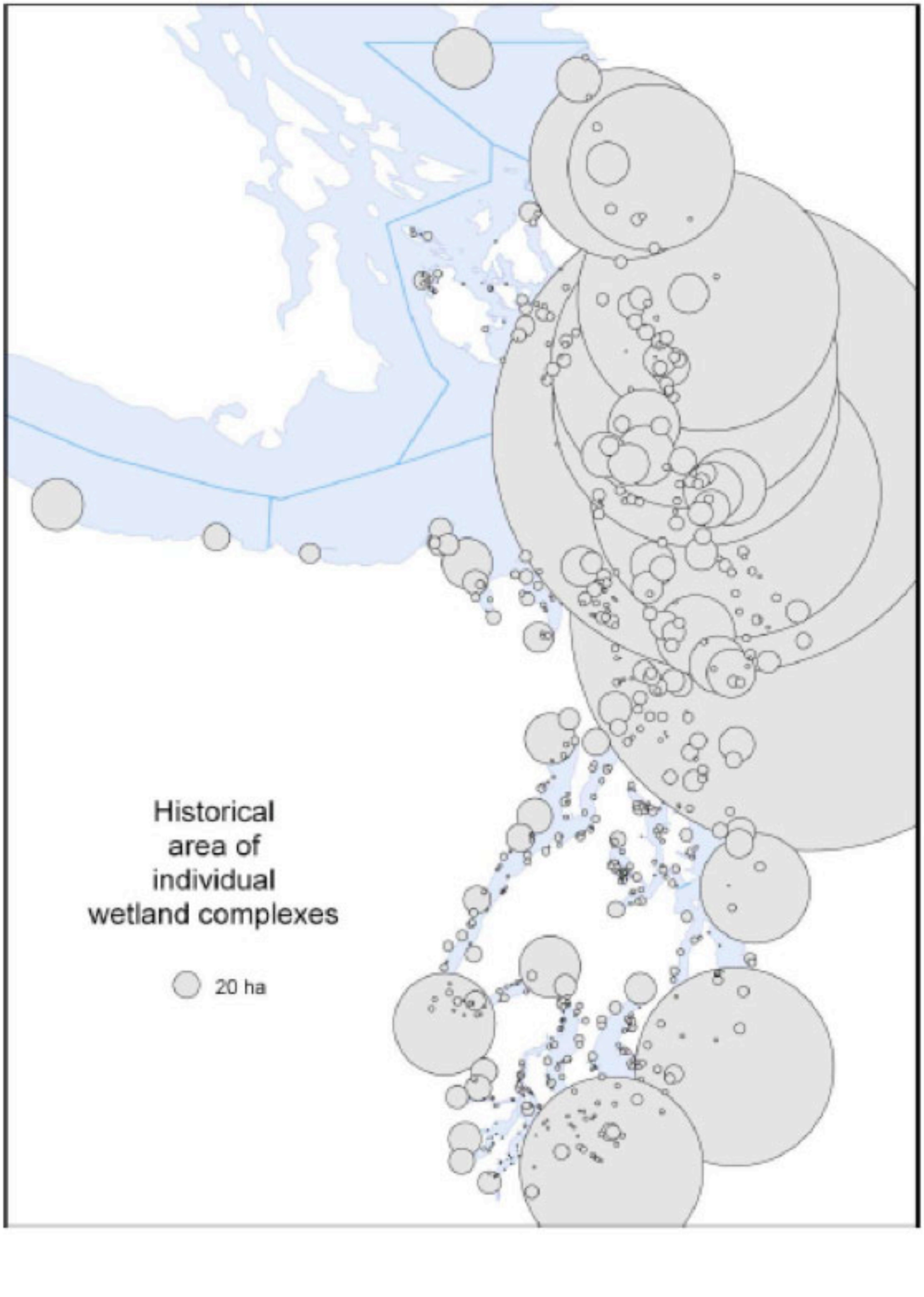


Figure 5. Historical area of individual wetland complexes (note: in all pie diagrams, wetland is proportional to the symbol area (reprinted with permission from Collins and Sheikh 2005)

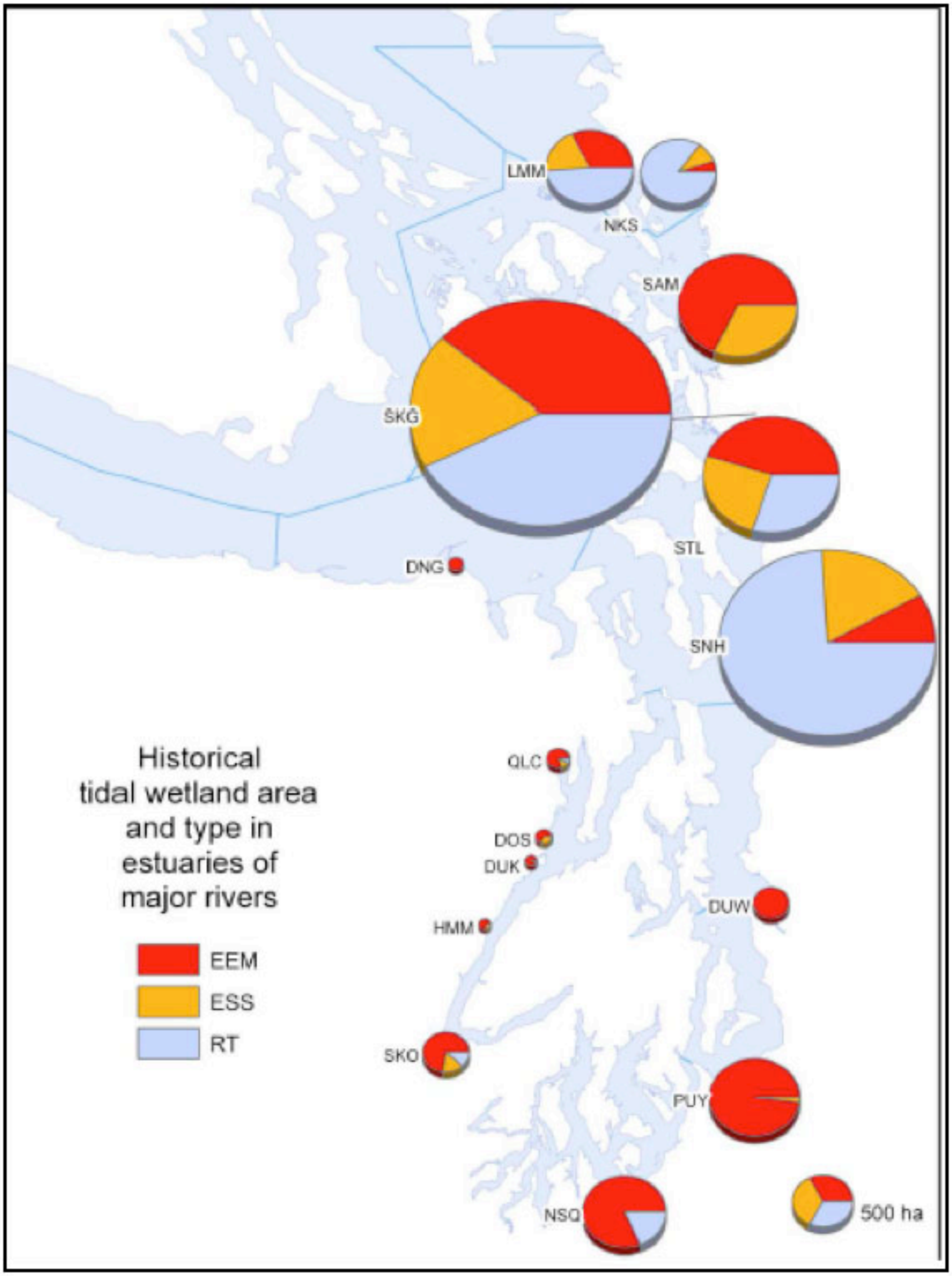


Figure 6. Relative area of historical tidal wetland types in the estuaries of major rivers draining the Cascade Range and Olympic Mountains. EEM- estuarine emergent wetland; ESS- estuarine

scrub-shrub wetland; RT- riverine-tidal wetland. (note: in all pie diagrams, wetland is proportional to the symbol area) (reprinted with permission from Collins and Sheikh 2005)

Uncertainties

Assessing the degree to which wetlands have changed over time is challenging. As with any analysis of historical trends, the frame of reference (baseline) can dictate the perception of change (e.g., Jackson et al. 2001), yet historical accounts are often less quantitative and thereby more subjective (Thom and Hallum 1990, Collins and Sheikh 2005). The use of historic maps from different sources is hindered by differences in terminology with respect to classifications of wetland hydrology, habitat or vegetation. Despite these challenges, the current efforts to recreate a quantitative picture of the extent and nature of historic wetlands have taken substantial measures to account for these difficulties (Thom and Hallum 1990, Collins and Sheikh 2005). The similarity of independent estimates derived from disparate sources strengthens confidence in them. Both Thom Hallum (1990) and Collins and Sheikh (2005) acknowledge that their estimates of historic wetland area may still be lower than their true extent given the limitations in the available data. The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP) is currently conducting a closer investigation of intertidal wetlands using the database created by WDNR and UW. This effort is ongoing and will likely yield a more detailed analysis of wetland change in Puget Sound. While there has been much recent and ongoing efforts to restore wetlands in Puget Sound, the effectiveness and long-term sustainability has not been determined for the entire Puget Sound, though monitoring programs are used to document progress towards this end. The existing comparisons between current and historic wetlands do not currently separate restored wetlands from natural ones.

Summary

Tidal wetlands play an integral role in the hydrology, chemistry and nearshore ecosystem of Puget Sound and have experienced significant declines as a result of industrial uses, agricultural uses, and other types of human development. While much of the wetland loss and alteration occurred after 1900, dredging and channeling of large river deltas began as early as the 1850s. There have been several investigations into wetland change since pre-industrial times, each utilizing divergent or common data sources and deriving generally consistent estimates. The most recent and comprehensive assessment documents that the current area of tidal wetlands in Puget Sound is 17-19 % of historic levels and that most of the loss has occurred in the Whidbey Basin (Collins and Sheikh 2005). Ongoing investigations by PSNERP stand to shed more light on the extent and nature of current and historic wetland alterations in Puget Sound. Currently, efforts to restore estuarine and tidal wetlands hold promise for recovering lost ecosystem function.

Literature Cited

Bortleson, G. C., M. Chrzastowski, and A. K. Helgerson. 1980. Historical changes of shoreline and wetland at eleven major deltas in the Puget Sound region, Washington. U.S. Geological Survey, U. S. Dept. of, Justice, U.S. Bureau of Indian, Affairs, Reston, Va.

Boule, M. 1981. Tidal wetlands of the Puget Sound region, Washington. Wetlands 1:47-60.

Boule, M. E., N. Olmsted, and T. Miller. 1983. Inventory of wetland resources and evaluation of wetland management in western Washington. The Dept. of Ecology, Olympia, WA.

Collins, B. D., D. R. Montgomery, and A. J. Sheikh. 2003. Reconstructing the historical riverine landscape of the Puget Lowland Pages 79-128 in D. R. Montgomery, S. M. Bolton, D. B. Booth, and L. Wall, editors. Restoration of Puget Sound Rivers. University of Washington Press, Seattle, WA.

Collins, B. D., and A. J. Sheikh. 2005. Historical reconstruction, classification, and change analysis of Puget Sound tidal marshes. University of Washington (Seattle, WA) and the Nearshore Habitat Program, Washington State Dept. of Natural Resources, Olympia, WA.

Eissinger, A. M. 2007 Great Blue Herons in Puget Sound. . Seattle, WA.

Jackson, J. B. C., M. X. Kirby, W. H. Berger, K. A. Bjorndal, L. W. Botsford, B. J. Bourque, R. H. Bradbury, R. Cooke, J. Erlandson, J. A. Estes, T. P. Hughes, S. Kidwell, C. B. Lange, H. S. Lenihan, J. M. Pandolfi, C. H. Peterson, R. S. Steneck, M. J. Tegner, and R. R. Warner. 2001. Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science 293:629-637.

McMillan, R. O., D. A. Armstrong, and P. A. Dinnel. 1995. Comparison of intertidal habitat use and growth rates of two northern Puget Sound cohorts of 0+ age Dungeness crab, Cancer magister Estuaries 18:390-398.

Mitsch, W. J., and J. G. Gosselink. 2007. Wetlands. 4th edition. John Wiley & Sons, Inc., Hoboken, NJ.

Nesbit, D. M. 1885. Tide marshes of the United States. U.S. Dept. of Agriculture, Washington, D.C.

Simenstad, C. A., and J. R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecological Engineering 15:283-302.

Stick, K. C., and A. Lindquist. 2009. 2008 Washington State Herring Stock Status Report. Stock Status Report No. FPA 09-05, Olympia, WA.

Thom, R. M., and L. Hallum. 1990. Long-term changes in the areal extent of tidal marshes, eelgrass meadows and kelp forests of Puget Sound. Wetland Ecosystem Team, Fisheries Research Institute, School of Fisheries, University of Washington, Seattle, WA.

Water Quality

Puget Sound is unique in the lower 48 Unites States because of its fjord-like physiography, inland extent, wide range of depths, and urbanized watersheds and shorelines. Limited exchange of seawater between sub-basins within Puget Sound can result in long residence times, potentially increasing the susceptibility of biota to contamination introduced through human activities. The varied habitats within Puget Sound support multiple life history stages of many species, potentially exposing sensitive life stages to contamination. There are multiple water quality concerns in Puget Sound:

- Levels of toxic contaminants in biota that live or feed in Puget Sound.
- The eutrophication of marine waters, producing hypoxic and anoxic regions.
- Wastewater contamination, principally from combined sewer overflows or septic systems
- Harmful algal blooms, which introduce toxins that enter the food web
- Acidification of marine waters, and the adverse ecological effects that result.

Degradation of water quality in Puget Sound occurs through three primary mechanisms. The first is through the introduction of toxic contaminants, primarily comprising manufactured synthetic chemicals, but also including compounds that occur naturally that are concentrated in the local environment to toxic levels via human activities. The second is through human-caused changes in naturally occurring chemicals, compounds, or physical parameters (e.g., temperature, turbidity, nutrients, pH). The third is through introduction of new diseases or pathogens, or through other activities that cause an unnatural increase in disease organisms.

Here we treat the these first two of these mechanisms, focusing on the marine and estuarine waters of Puget Sound, and restricting our treatment to degradation caused by human activities. Future editions of the Update will expand the treatment to include pathogens, the condition of fresh water systems, and natural sources of change in water quality.

1. Toxic Contaminants

Background

Determination of the significance of contamination of the Puget Sound ecosystem by toxic chemicals requires measuring the health of organisms, understanding how toxics move through the ecosystem, and estimating the risks posed by exposure to toxic chemicals. In this report we integrate some of the physical characteristics of toxics in the system with the negative effects they could cause on biota. The “threat” of toxics is dealt with separately in Section 3. Here we provide a comprehensive overview of toxics in the system, regardless of their value as an indicator of water quality. Thus, some information presented in this section comes from metrics that may not be the best indicators of water quality, but instead addresses issues of human or ecosystem health (e.g., salmon).

Toxic contaminants have been released into the Puget Sound and its watersheds for decades by human activities. Concern over the possible harmful effects of these pollutants in the ecosystem led to the creation of Washington’s Pollution Control Commission in 1945, almost 30 years before the federal Clean Water Act. The Puget Sound Water Quality Authority was established in 1985 to address pressing water quality issues, and by 1989 monitoring and assessment of water quality in Puget Sound had begun with the Puget Sound Ambient Monitoring Program (PSAMP).

The goals of PSAMP included characterizing status and trends of the condition of Puget Sound. Now called the Puget Sound Assessment and Monitoring Program, it currently exists as a consortium of regional scientists from a number of agencies who monitor and assess ecosystem health. Other ongoing toxics monitoring efforts in Puget Sound include MusselWatch (Kimbrough et al. 2008), a national program that has been active in Puget Sound since the 1980s, and King County’s Marine Monitoring Program .

The Washington Department of Ecology has evaluated and identified 17 chemicals of concern for Puget Sound (Table 1)(Hart Crowser 2007), based on threat or known harm to biota. Of these, only five chemicals have been banned nation-wide under the Toxic Substances Control Act (TSCA) since 1976. Washington State recently became the first state to ban a class of relatively new chemicals, polybrominated flame retardants (PBDEs), because of human and environmental health concerns.

Table 1. Washington Department of Ecology’s list of Chemicals of Concern. Table reprinted from Hart Crowser 2007

Chemical of Concern	Category Addressed	Harm or threat
Arsenic	Arsenic	Associated with sediment toxicity and benthic community impairment
Cadmium	Cadmium	Accumulation in shellfish
Copper	Copper	Associated with sediment toxicity and benthic community impairment; affects salmonids and stream health
Lead	Lead	Associated with sediment toxicity and benthic community impairment
Mercury	Mercury	Target of fish consumption advice; Associated with sediment toxicity and benthic community impairment
Total PCBs (a)	PCBs	Target of fish consumption advice; accumulation in fish, birds, mammals; associated with sediment toxicity and benthic community impairment
Low molecular weight PAHs (b)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Carcinogenic PAHs (c)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Other high molecular weight PAHs (d)	PAHs	Liver lesions and reproductive impairment in fish from urban bays; associated with sediment toxicity and benthic community impairment
Sum of DDT and metabolites (e)	Pesticides	Accumulation in fish, birds, and mammals; associated with sediment toxicity and benthic community impairment
Triclopyr (f)	Pesticides	Category thought to affect salmonids and stream health
Total dioxin TEQs from dioxins & furans (g)	Dioxins and furans	Accumulation in birds and mammals; furans associated with sediment toxicity and benthic community impairment
bis(2-Ethylhexyl)phthalate	Phthalate esters	Category shown to accumulate in fish, invertebrates, and sediment of urban waterways at levels triggering sediment clean up activities
Total PBDEs (h)	PBDEs	Accumulation in sediments, fish, and harbor seals
Nonylphenol	Hormone disrupting chemicals	Category thought to cause reproductive impairment observed in fish from urban bays
Oil or petroleum product (i)		Kills and reduces fitness of marine organisms
Zinc		Increasing concentrations may threaten aquatic resources

(a) Sum of polychlorinated biphenyl congeners.
(b) Polyaromatic hydrocarbons: acenaphthene, acenaphthylene, anthracene, fluorene, naphthalene, and phenanthrene (per WAC 173-204-320).
(c) Polyaromatic hydrocarbons: benz(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, benzo(k)fluoranthene, chrysene, dibenz(a,h)anthracene, and indeno(1,2,3-c,d)pyrene (per USEPA).
(d) Polyaromatic hydrocarbons: benzo(g,h,i)perylene, fluoranthene, and pyrene (WAC 173-204-320 high molecular weight PAHs not on U.S. EPA list of carcinogenic PAHs).
(e) DDT = Dichlorodiphenyltrichloroethane.
(f) Input from the project team did not reflect consensus to include this compound as currently used pesticide. Other candidates suggested by project team members included diazinon and dichlorbenil.
(g) TEQ = Toxicity equivalent.
(h) PBDEs = Polybrominated diphenyl ethers. Sum of congeners have been normalized.
(i) Specified as crude oil, specific refined product (e.g., diesel, gasoline, heavy fuel oil), or analytical result as TPH-D or TRPH.

Toxic contaminants are considered a priority threat in Puget Sound because they may harm the health of biota. In many cases harm can be difficult to observe; effects can be non-lethal (behavioral) or affect reproductive potential. The status of toxic contaminants in ecosystems typically is reported using a) metrics of exposure, such as the concentration of contaminant residues in tissues; b) health effects such as cancer or reproductive impairment that are known to be caused by such exposure (i.e., are "toxicopathic"); c) concentration of toxics in abiotic media such as sediments or water; d) toxicity of abiotic media; e) benthic infaunal community metrics, or f) an index value calculated from some combination of a-e. The process of "bioconcentration" of toxics from abiotic media to biota is well documented in some cases, suggesting that toxic contaminants in abiotic media can serve as a proxy for or predictor of exposure (Meador 2006).

Measuring toxic contaminants in the environment is expensive and sometimes logistically difficult, so monitoring and assessment studies or programs are challenged with targeting contaminants that pose the greatest threat based on a number of criteria including:

- level of toxicity to organisms
- types of harm caused
- persistence in the environment
- rates of bioaccumulation and biomagnification
- frequency of occurrence in the ecosystem
- spatial distribution in the ecosystem
- threats to specific taxa

Furthermore, the toxicity of a contaminant to an organism depends on the degree to which it is exposed to the chemical. Ideally, status is reported with respect to both the degree of exposure, and the effects (impacts) that exposure causes.

This section summarizes the status and trends of contaminant exposure and effects for key species to four major classes of toxic contaminant. Metrics reported here include: a) measurements of contaminant concentration in organisms' bodies ("tissue residues") or concentration of contaminant metabolites; b) toxicopathic effects (e.g., liver disease and various measures of reproductive impairment); c) concentration of toxics in sediments, primarily as a source of and proxy for biotic exposure; and d) a multimetric toxics-related index of sediment health.

The focus of this section is on toxic contaminants as they relate to biotic exposure and effects. Various species have been used over the years as indicators of toxics status and trends, based on key life history characteristics designed to evaluate the presence, fate, and transport of toxics in the complete food web. Key life history characteristics include:

- Where the organism lives (its habitat, e.g., benthic vs pelagic)
- Trophic level
- Longevity (long lived species have a greater potential for accumulative exposure)
- Migration/residency relative to contaminated habitats
- Prey or food preferences

Furthermore, the focus of this report is limited to the marine ecosystem. Evaluation of loadings and sources, such as from stormwater or atmospheric deposition, is not included.

Toxic contaminants in sediments and fish tissues have been two of the most widely monitored and assessed indicators of ecosystem health in Puget Sound. Understanding the significance of the threat posed by sediment contamination requires an understanding of the relationship between sediment pollution, biotic exposure, and the movement of contaminants from sediments to biota. The majority of data useful for a broad-scale evaluation of status and trends in both sediment and biota comes from the PSAMP long-term monitoring and assessment studies. Results from these efforts have been published primarily in the periodic Puget Sound Update series and in other state agency reports. Most PSAMP data collection methods use vetted protocols (e.g., Puget Sound Estuary Program 1989a (revised), 1989b (Revised), 1990, 1996a, 1996b) which may have been modified over time following internal agency peer review or review among PSAMP principal investigators. Reviews of PSAMP were performed by a panel of external experts in 1995 (Shen 1995) and again in 2005 by PSAMP's Management Committee (PSAMP unpublished). In cases where agency-endorsed or other adequate processes for peer review were performed, and where procedures were vetted as above, PSAMP results from the Puget Sound Update series or other Agency reports are cited or presented here. Data or findings that fail to meet these requirements are omitted.

Status and Trends

Persistent Bioaccumulative Toxics (PBTs)

Persistent bioaccumulative toxic contaminants are a class of substances comprising primarily synthetic chemicals designed and manufactured to meet a wide range of industrial, agricultural, or residential needs. Because they are persistent and bio-accumulative, they are cause for concern when released into the environment. These chemicals generally resist physical, chemical, and metabolic breakdown, so they remain unchanged in the environment for a long period of time. Their concentration increases in the body with chronic or increasing exposure or intake, and they are toxic, causing harm to biota. Because of these characteristics, PBTs have been the focus of intense research world-wide, and large PBT databases exist for risk assessors, modelers, and regulators (Weisbrod et al. 2007).

In Puget Sound marine and estuarine waters the PBTs of primary concern are summarized by Hart Crowser (2007). Those for which broad status information exists in Puget Sound include polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), organo-chlorinated pesticides (OCPs) such as dichlorodiphenyltrichloroethane (DDTs), and mercury. These contaminants have been measured or monitored in a wide range of species in Puget Sound from as early as the mid 1970s to present, with consistent monitoring in several species from 1989 to present. Although polychlorinated dibenzo-p-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) have been detected in English sole from the most heavily contaminated embayment in Puget Sound (Elliott Bay; Sloan and Gries 2008), these compounds are considered a minor threat to apex predators such as harbor seals in Puget Sound (Ross et al. 2004) that could otherwise potentially be exposed to high PBT levels via bio-magnification.

Perhaps the clearest PBT exposure-effects relationship of concern in the Puget Sound marine waters is the exposure of apex predators such as Southern Resident Killer Whales (SRKW) and harbor seals to PCBs and PBDEs (Figure 1). Hickie et al. (2007) and Ross et al. (2004) reported PCB exposure in harbor seals from Puget Sound at levels predicted to impair health, while Ross et al. (2000) described the SRKW population as the most PCB-contaminated of all cetaceans in the world. Calculations made by Hickie et al. (2007) and Ross et al. (2004) suggested that during their years of peak exposure, all members of the SRKW population were affected, and that exposure exceeded thresholds by 3 to 31 times. The authors estimated that based on PCBs alone, it would take until the year 2089 for 95% of the population to drop below the health effects threshold, given current PCB trends. Such PBT contamination is considered a risk to recovery of this population (Krahn et al. 2002).

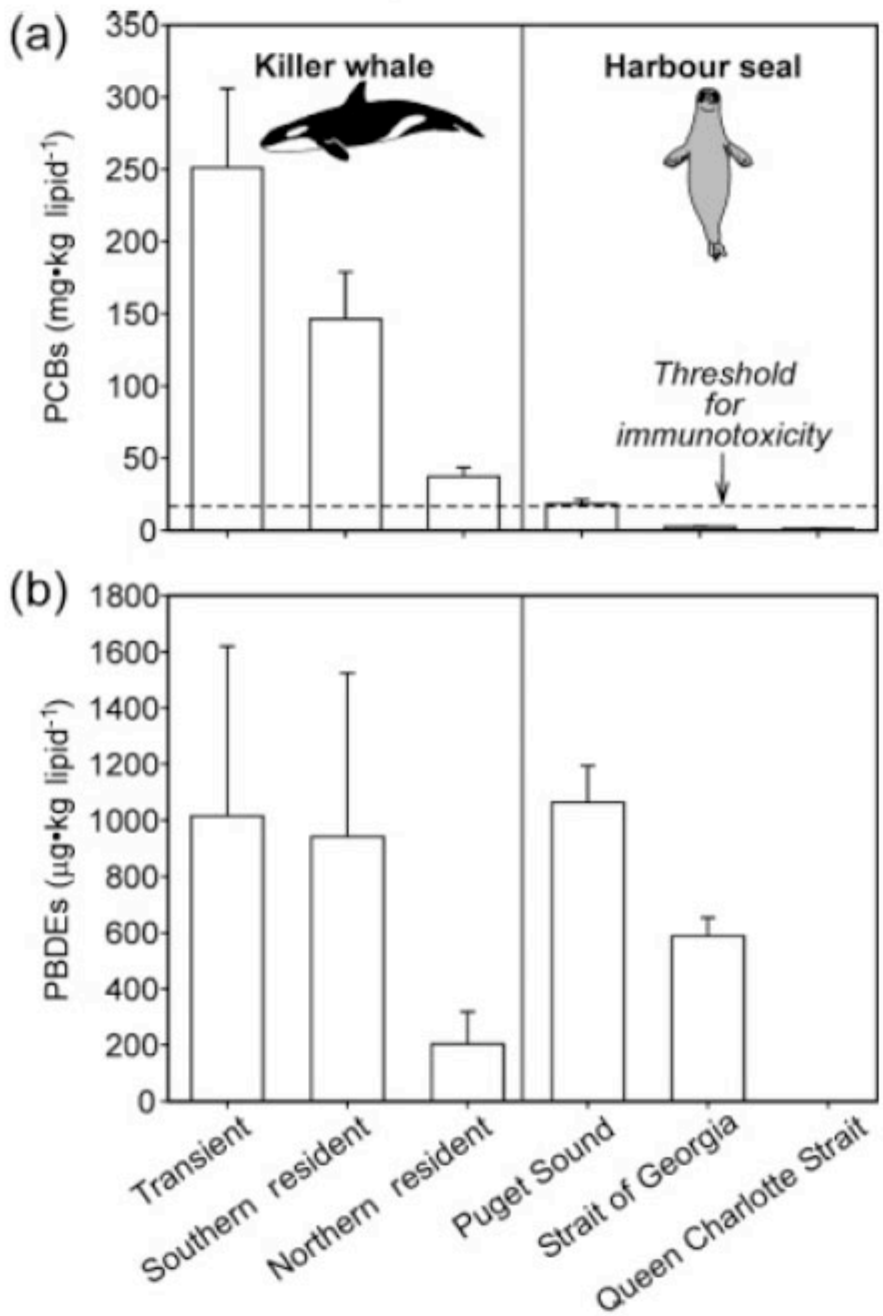


Figure 1. Persistent bioaccumulative toxics (PCBs and PBDEs) in two apex predators from the Puget Sound and Strait of Georgia, with health effects threshold for PCBs. Reprinted with permission from Ross (2006)

The source of PCBs to these animals is their food, primarily chinook salmon for killer whales (e.g., Krahn et al. 2007) and a mix of small pelagic and benthic fish for harbor seals (Cullon et al. 2005). O'Neill and West (2009) reported high PCB body burdens in chinook salmon that reside in Puget Sound, compared to more oceanic migrants (Figure 2) and West et al. (2008) reported high PBC burdens in Pacific herring from Central and Southern Puget Sound, compared with Southern Strait of Georgia and with herring from highly polluted regions of the Baltic Sea (Figure 3). This illustrates the importance of PBT transfer via trophic interactions and the need to understand PBT fate and transport processes in food webs.

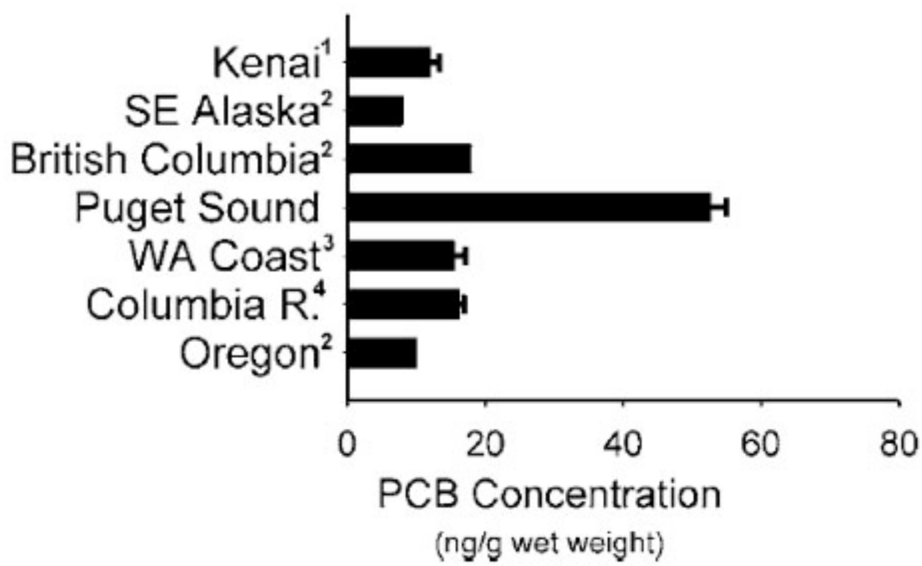


Figure 2. Comparison of PCB tissue residues in adultChinook salmon returning to spawn in Puget Sound and Pacific Oceanic coastal rivers. See West and O’Neill 2009 for a description of sampling location and full data citations. All samples were from adult Chinook salmon returning to natal rivers to spawn. Copyright American Fisheries Society. Used with permission.

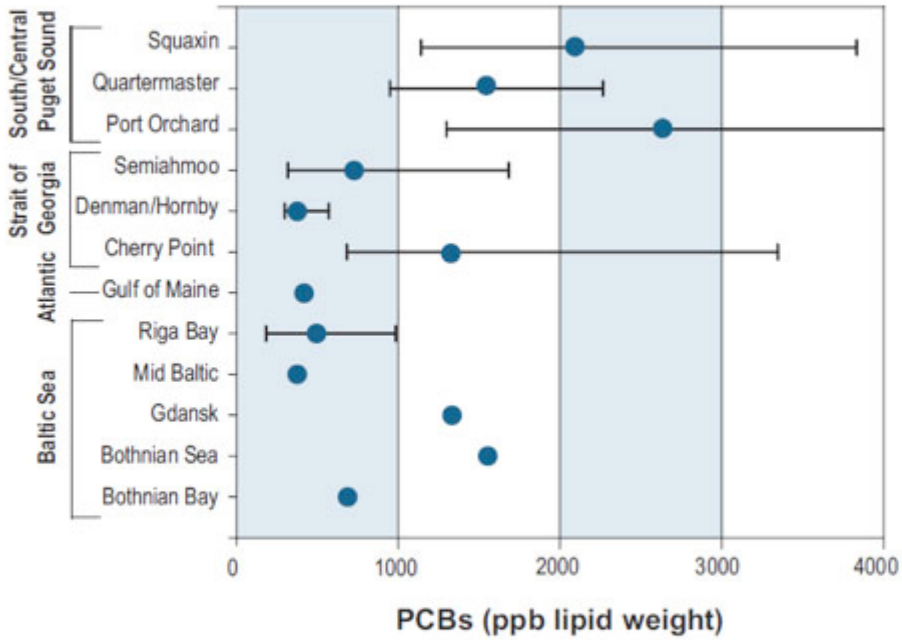


Figure 3. Comparison of PCBs among six populations of Pacific herring Puget Sound and the Georgia Basin, and Atlantic and Baltic herring. Squaxin population from South Puget Sound, Quartermaster Harbor and Port Orchard from Cental Puget Sound, and Semiahmoo, Denman/Hornby, and Cherry Point from the Southern Strait of Georgia (Reprinted from 2007 Puget Sound Update; data from West et al. 2008)

PCB exposures in chinook salmon pose a health risk to the fish, as well as to the humans that consume them. PCBs in chinook salmon(O'Neill and West 2009) exceeded an effects threshold reported by Meador (2002), indicating a threat to normal growth and maturation processes for these salmon. Furthermore, the Washington Department of Health has issued guidelines that recommend restrictions to dietary intake of these fish to protect human health .

In sediments, PBTs tend to accumulate in industrial or urbanized habitats near their sources, prompting focused attention on toxics there (Partridge et al. 2009, Puget Sound Estuary Program 1988). The Environmental Protection Agency's Superfund program has focused sediment cleanup efforts in a number of Puget Sound's urbanized embayments since 1980 (2007 Puget Sound Update). Overall, however, Ecology's long-term PSAMP efforts (methods peer reviewed: Dutch et al. 2009) have reported PCB levels in sediments exceeding Washington State Sediment Quality Standards (SQS) in only 19 of over 500 stations from the full extent of Puget Sound sampled between 1997 and 2008. Washington State Sediment Quality Standards, adopted as part of Washington's environmental regulations, define levels at which various chemicals present in sediments become harmful to marine life (WAC 173-204). All PCB exceedances were located in sediments taken from urban embayments in the Central Puget Sound basin. Data indicate that PCB concentrations in Elliott Bay sediments, where most of the exceedances have occurred, have been declining (Partridge et al. 2009). A Washington State Sediment Quality Standard does not yet exist for PBDEs, and although PBDE concentrations were lower than PCBs overall, they were concentrated in Central Puget Sound and its urbanized embayments.

Long-term Sound-wide monitoring efforts have shown that this urban PCB and PBDE sediment signal is reflected in benthic (bottom-dwelling) and demersal (near-bottom) species. Tissue residues of PCBs and PBDEs were greatest in English sole (benthic), rockfish (demersal) and lingcod (demersal) from Elliott Bay, Commencement Bay, and Sinclair Inlet, or from other Central Puget Sound urban or near-urban locations (as reported in 2007 Puget Sound Update). PCB residues in blue mussels were greatest in Central Puget Sound locations (Kimbrough et al. 2008). These studies demonstrate the relationship between benthic (or benthic-feeding) species and the contaminant-condition of their environment.

Although pelagic (open-water) species may not have direct trophic connections with the sediment-contaminated benthic food web, pelagic food web species in urbanized waters exhibited high levels of exposure to PBTs. Pacific herring (West et al. 2008), Chapter2a.Salmonids#chinookanchor|chinook salmon]] (O'Neill and West 2009), and harbor seals (Ross et al. 2004) that reside in Puget Sound conform to this pattern. PCB and PBDE tissue residues were consistently greatest in individuals of these three species from the Central or Southern Puget Sound Basins, compared with conspecifics from the Strait of Georgia, Strait of Juan de Fuca, or Pacific Ocean. As noted previously, PCB and PBDE tissue residues exceeded health effects thresholds in salmon] and [[Chapter2a.HarborSeals|harbor seals.

PBDEs have only relatively recently been added to tissue monitoring and assessment programs in Puget Sound. Using archived tissue samples, West and O'Neill (2007) observed 80 ng/g Total PBDEs (wet wt) in herring from Central Puget Sound in 2001, roughly one-half the concentration of Total PCBs reported for the same samples from (West et al. 2008).

Polycyclic Aromatic Hydrocarbons (PAHs)

PAHs are derived from fossil fuels, and are typically produced via combustion of these fuels (pyrogenic) or occur as constituents of petroleum (petrogenic). Most of these chemicals exist naturally, but their presence in the environment becomes problematic when they are concentrated to toxic levels by human activities. Many PAHs are persistent in the environment, however they are typically metabolized by vertebrates when exposed to relatively low concentrations, and therefore do not tend to accumulate in their bodies. For this reason, food web magnification of PAHs for apex predators is of less concern than for PBTs.

However, both exposure and effects measures from biota indicate that PAHs represent a serious threat to the health of some Puget Sound biota. PAHs in blue mussels from seven of 14 sites in Puget Sound waters were termed “high” (at or above the 85th percentile for all 263 stations nationwide in at least half the years sampled between 1986 and 1991) by the national Mussel Watch Program (O'Connor 2002). Currently, the PAH status of mussels from eight of 10 stations in Puget Sound is rated either “medium” or “high” (Kimbrough et al. 2008), with a number of locations that met or exceeded comparable mussel samples taken in highly urbanized areas of the Baltic Sea. Tissue residues of PAHs in blue mussels could originate from capturing and consuming PAH-laden particles derived from nearby sediments (Baumard et al. 1999). This hypothesis is supported by the observation that PAHs in Puget Sound mussels are typically greatest in urban sites (Kimbrough et al. 2008).

Because PAHs are metabolized by vertebrates, measuring the exposure of fish, birds and mammals is less straightforward than measuring PBT tissue residues. Metabolites of PAH compounds can be measured in fish bile (Krahn et al. 1984), and these so-called biliary Fluorescing Aromatic Compounds (FACs) have been monitored in English sole, rockfish and herring as a semi-quantitative measure of PAH exposure in these species in Puget Sound (West et al. 2001). In the benthic or demersal fish species biliary FACs were consistently greatest in fish taken from urbanized embayments.

PAHs are linked to a number of toxicopathic fish diseases. English sole develop degenerative liver disease when exposed to PAHs in the sediments where they feed (Myers et al. 1990, Myers et al. 1991). Other effects include interruption in growth, and reproductive impairments (Johnson et al. 2002). Myers et al. (2008) documented the recovery of health among English sole in Eagle Harbor, a highly PAH-contaminated Superfund site, where prevalence of PAH-induced liver disease dropped from 80% to 5% over a ten year period during remediation, which included sequestration of PAHs with sediment capping (Figure 4).

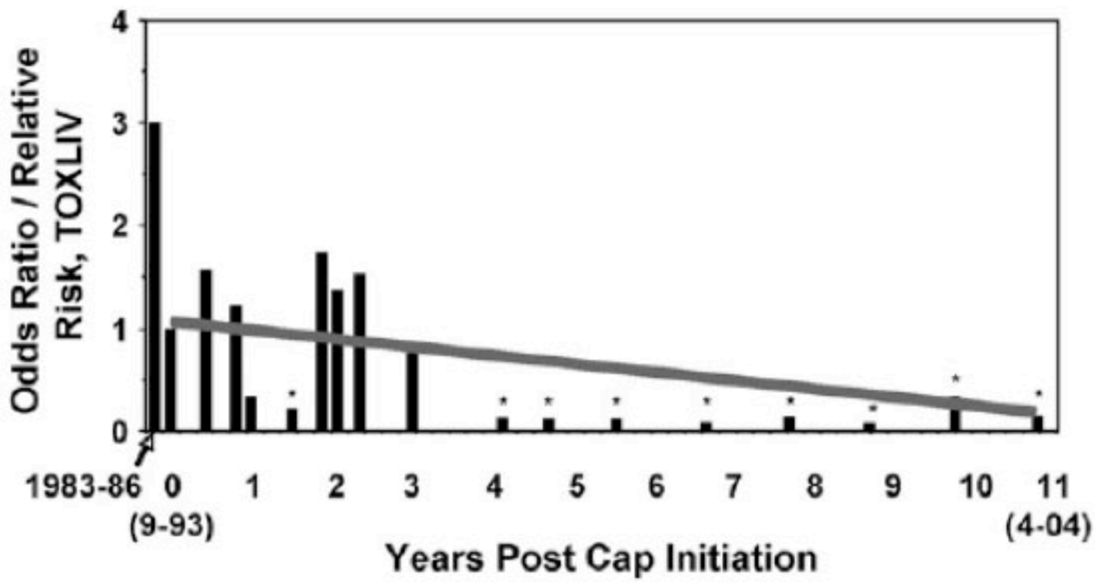


Figure 4. Temporal trend in liver disease of English sole from a sediment-remediated site in Eagle Harbor, Washington. Reprinted with permission from Myers et al. 2008.

PAH-induced liver disease has been tracked in English sole by PSAMP for 20 years in eight Puget Sound locations. This tracking study uses protocols developed to monitor histopathological health metrics in fish, including toxicopathic liver disease (Puget Sound Estuary Program 1987), which are regularly reviewed by a contract pathologist. Recent results from this tracking study, reported in the 2007 Puget Sound Science Update, include the following:

- liver disease in English sole is associated primarily with Puget Sound's highly contaminated urban embayments near Seattle (Elliott Bay), Tacoma (Thea Foss Waterway), and Everett (Port Gardner). The risk of developing liver disease in these areas was two to six times that of normal background
- the risk of developing PAH-induced liver disease has remained low and unchanged at six of eight long-term stations, and has declined significantly in English sole from Elliott Bay (Seattle).
- high molecular weight PAHs, the group most often associated with liver disease (Myers et al. 1991), have declined in Elliott Bay sediments (Partridge et al. 2009), and in English sole bile from Elliott Bay

Pelagic fish in Puget Sound have also exhibited exposure to PAHs. Pacific herring, a small, schooling pelagic planktivorous fish, have consistently exhibited PAH exposures in Central Puget Sound similar in magnitude to benthic (English sole) and demersal (rockfish) species from most urbanized embayments for the past 10 years (2007 Puget Sound Science Update).

The source of persistent organic pollutants such as PAHs in adult fishes is widely thought to be dietary and because PAHs are metabolized, their biliary FAC measurements tend to reflect PAH loads in prey that have been consumed recently. Pacific herring is a fully pelagic species that consumes primarily zooplankton prey, with little obvious trophic connection to contaminated sediments in Puget Sound. Small schooling pelagic planktivores such as herring may accumulate PAHs that have originally been ad- or absorbed to plankton, and then magnified among planktonic invertebrates in the food web (Wolfe et al. 2001). It has been suggested that some PAHs loaded from atmospheric deposition or other sources enter the pelagic food web directly via bioaccumulation by phytoplankton, and then are magnified through the planktonic food chain to their fish predators (Larsson et al. 2000). In Puget Sound, this may explain why pelagic species far removed, both trophically and spatially, from contaminated sediments exhibit such high exposure to PAHs, and could help to inform decisions regarding how to mitigate exposure of biota to PAHs in Puget Sound.

PAH exposure may pose a significant threat to sensitive life stages of Puget Sound biota. Links between chronic, sublethal PAH levels and health impacts in fish embryo and larval stages, as well as delayed population declines from early-life PAH exposures have been well established (Carls and Meador 2009, Peterson et al. 2003). In addition, PAHs from creosote, such as found on treated pilings, can kill embryos (Vines et al. 2000). Herring embryos exhibiting chronic mortality from at least one spawning ground in Puget Sound have experienced exposure to PAHs exceeding a PAH effects threshold (as reported in 2007 Puget Sound Science Update), however a PAH cause-and-effect link to mortality has not yet been established in Puget Sound.

Contaminants of Emerging Concern (CECs)

This group of contaminants comprises a broad range of chemical classes whose adverse effects on biota is only recently becoming apparent. They range widely in solubility, persistence, toxicity, and mode-of-action, and include such classes as perfluorinated compounds (from the creation of fluoropolymers, semiconductors, and fire-fighting foam), nonylphenol (a surfactant), bisphenol-A and phthalates, both used in plastics, and pharmaceuticals and personal care products. Many of these contaminants have endocrine disrupting capacity, and so may be reported as Endocrine Disrupting Compounds (EDCs); some are specifically estrogenic and so may be reported as xenoestrogens. Although some of these contaminants have been measured in Puget Sound fishes (West et al. 2001), and some are monitored in Puget Sound sediments (Dutch et al. 2009), many CECs currently are not measured in environmental media on a broad scale (Muir and Howard 2006). Moreover, analytical techniques for measuring tissue residues or metabolites for many CECs are lacking. Two CECs, nonylphenol (NNP) and bis(2-ethylhexyl)phthalate (DEHP) are included in Washington Department of Ecology's Chemicals of Concern list (Hart Crowser 2007).

In Puget Sound sediments, at least one phthalate-chemical, DEHP, exceeded the Washington State sediment quality standard, and appears to be increasing (Partridge et al. 2009). DEHP is associated with a wide range of toxicopathic disease including endocrine disruption (e.g., feminization of males). English sole in Puget Sound have shown evidence of exposure to xenoestrogenic compounds, even though the causative pollutants remain unknown. Johnson et al. (2008) reported the presence of vitellogenin, a precursor to egg protein normally found only in females, in male English sole from twelve of sixteen locations sampled in Puget Sound. Moreover, both females and males from one affected population in Elliott Bay exhibited altered reproductive timing, possibly related to the unknown estrogenic pollutants (Figure 5). Based on spatial patterns in the fish impairment, these authors hypothesized that the source of xenoestrogens to these fish was industrial discharges, surface runoff, or sewage, and discussed the most likely causative pollutants.

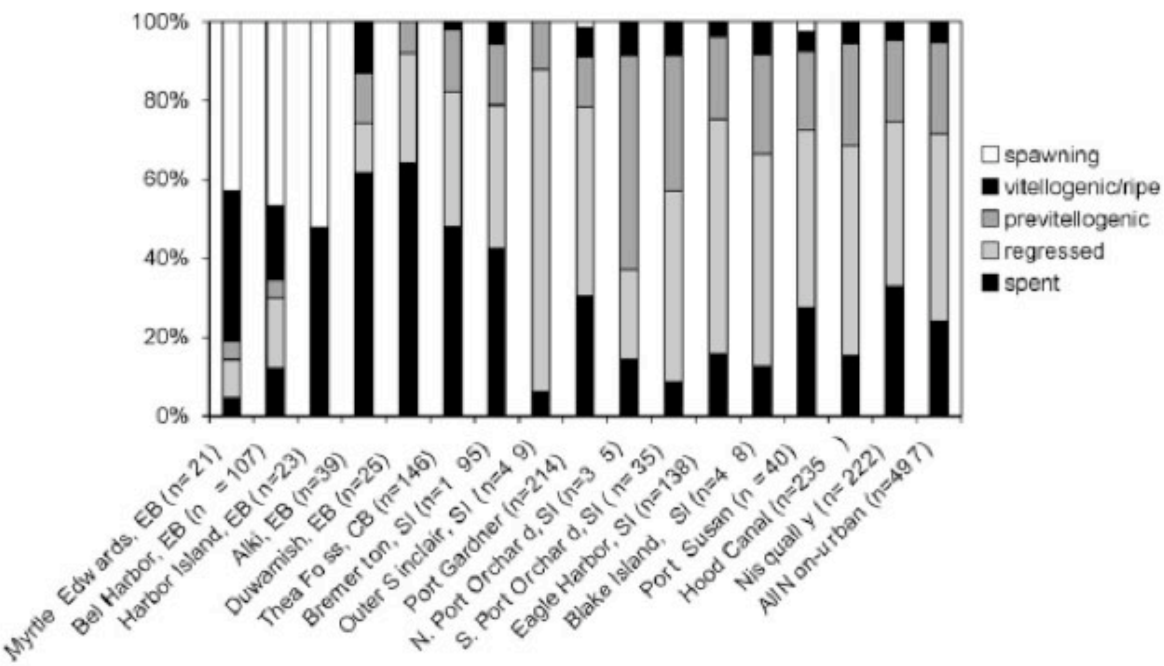


Figure 5. Unusual reproductive timing in female English sole from three Elliott Bay sites compared to 14 other Puget Sound locations. Reprinted with permission from Johnson et al. 2008.

Metals

Like PAHs, metals occur naturally in the environment. Metals become contaminants of concern when they are altered chemically or redistributed in the environment in ways that make them more available or toxic to biota. In some cases (e.g., mercury) metals may naturally occur in biota in a magnitude great enough to cause concern for humans consuming them (Barghigiani and DeRanieri 1992).

Metals have been monitored Sound-wide in sediments (Dutch et al. 2009) and fish tissue (West et al. 2001) since 1989, and in blue mussels since 1986 (Kimbrough et al. 2008). Metal accumulation in mussels has been unremarkable, except that the greatest tissue residues of mercury, nickel, and lead occurred in highly urbanized areas, suggesting anthropogenic contributions. "Medium" concentration of a number of metals was reported from locations with the greatest exposure to oceanic waters, far removed from human activities, suggesting accumulation of natural sources.

The Sediment Quality Triad

The Sediment Quality Triad Index (SQTI) is a multi-metric index developed to describe the degradation of sediment condition by toxic contaminants (Chapman 1990). Because the SQTI incorporates toxic contaminants from a broad range of classes, it is presented separately in this section. The SQTI combines the results from pollutant concentration measures, toxicity studies (exposing sensitive organisms to sediments or their extracts), and analysis of the composition of the infaunal community in sediments. This last measure is typically based on best professional judgment, and integrates nine different measures of community structure and the presence/absence of pollutant tolerant/sensitive species. A substantial advantage of the SQTI is that it examines both effects and exposure metrics, and uses a weight-of-evidence approach to integrate three important measures of sediment quality into one indicator that can be compared Puget Sound-wide.

SQTI has been used extensively in Puget Sound as an indicator of sediment health (Long et al. 2003). A seven-year comprehensive SQTI survey of 381 Puget Sound sediment stations reported that although only one percent of sediments were "degraded", most of these sediments were in highly productive shallow-water embayments or river deltas (reported in 2007 Puget Sound Science Update). Thirty-eight percent of sediments in Puget Sound were considered of "intermediate" quality, wherein degradation was detected in one or two of the three SQTI metrics. A full, final report for this study is currently being reviewed by the Washington Department of Ecology.

Uncertainties.

One important uncertainty concerns the linkage between the status and trends of toxic contaminants in the ecosystem and the associated population-level effects on biota. Health-effects thresholds are lacking for the great majority of toxic contaminants monitored in Puget Sound, especially for complex mixtures of chemicals. Constructing models that predict population-level effects from lethal or sub-lethal effects of single contaminants or mixtures is a challenge, because such models often carry a great deal of uncertainty that can result in wide-ranging outcomes. Except for oil spills or other episodic events, observations of mortality directly attributable to toxic contaminants are rare. A singular exception to this in Puget Sound is pre-spawning mortality of coho salmon in urban streams (Collier et al. 2004), a phenomenon widely attributable to road-based contaminants. In this case Spromberg and Scholz (2009) predict extirpation of coho spawning runs over decadal time scales in urbanized streams.

In addition, recent findings on the susceptibility of eggs and larvae of Pacific herring to fossil fuel-derived PAH compounds (Carls et al. 1999, Incardona et al. 2009) combined with field studies that demonstrate exposure of their embryos to PAHs in Puget Sound (as reported in 2007 Puget Sound Update) show the risk of mortality to this sensitive life stage from exposure to chemical pollutants in Puget Sound. However, because such mortalities are extremely difficult to observe or measure in the wild, they currently are not used in decision-making.

Uncertainties unique to status and trends of monitoring data include shifting methodologies and study designs over long time periods. For example, PCBs reported from some studies in this document have been analyzed using a range of methodologies including mixtures (Aroclor) analysis and congener-based methods. Careful evaluation of all methods, including those for biological covariates, must be made when comparing these data across studies, and when applying threshold criteria.

Summary

Human activities have resulted in the introduction or elevation of toxic contaminants into Puget Sound. These include Persistent Bio-accumulative Toxics such as PCBs, PBDEs and DDTs, chemicals derived from fossil-fuels (PAHs), various metals, and Contaminants of Emerging Concern, including Endocrine Disrupting Compounds and pharmaceuticals and personal care products. In Puget Sound, a number of PBT chemicals are present in apex predators such killer as whales and harbor seals and in their primary food sources (salmon and herring) in concentrations that may harm their health and impair recovery of populations that are depressed. For most PBTs, the highest concentrations occur in sediments and biota from the Central Puget Sound and its urbanized embayments, or localized urbanized shorelines in other Puget Sound basins. PAH monitoring of mussel tissue has caused Puget Sound to be characterized as a hot spot for that class of contaminants, relative to other urban areas in the nation. PAH chemicals have also been detected in fish bile and identified as a causative factor in liver disease in English sole in Puget Sound waters. Juvenile life stages of fish may be particularly susceptible to PAH toxicity. Reproductive effects of endocrine-disrupting compounds have been detected in benthic Puget Sound fish but the consequences of exposure at the population level and long-term trends are not known.

Literature Cited

Barghigiani, C., and DeRanieri, S. 1992. Mercury content in different size classes of important edible species of the Northern Tyrrhenian Sea. Marine Pollution Bulletin 24(2): 114-116.

Baumard, P., Budzinski, H., Garrigues, P., Narbonne, J.F., Burgeot, T., Michel, X., and Bellocq, J. 1999. Polycyclic aromatic hydrocarbon (PAH) burden of mussels (*Mytilus* sp.) in different marine environments in relation with sediment PAH contamination, and bioavailability. Marine Environmental Research 47(5): 415-439.

Carls, M.G., and Meador, J.P. 2009. A Perspective on the toxicity of petrogenic PAHs to developing fish embryos related to environmental chemistry. Human and Ecological Risk Assessment: An International Journal 15(6): 1084 - 1098.

Carls, M.G., Rice, S.D., and Hose, J.E. 1999. Sensitivity of fish embryos to weathered crude oil: Part I. Low-level exposure during incubation causes malformations, genetic damage, and mortality in larval Pacific herring (*Clupea pallasi*). Environmental Toxicology and Chemistry 18(3): 481-493.

Chapman, P.M. 1990. The sediment quality triad approach to determining pollution-induced degradation. Science of The Total Environment 97-98: 815-825.

Collier, T.K., Labenia, J., Myers, M.S., and Scholz, N.L. 2004. Abstracts from the twelfth international symposium on pollutant respsonses in marine organisms - mechanisms of toxicity - organics: Extreme PRIMO: premature mortality of adult coho salmon associated with stormwater discharges. Marine Environmental Research 58(2-5): 359-369.

Cullon, D.L., Jeffries, S.J., and Ross, P.S. 2005. Persistent organic pollutants in the diet of harbor seals (Phoca vitulina) inhabiting Puget Sound, Washington (USA), and the Strait of Georgia, British Columbia (Canada): a food basket approach. Environmental Toxicology and Chemistry 24(10): 2562-2572.

Dutch, M., Partridge, V., Weakland, S., Welch, K., and Long, E. 2009. Quality Assurance

Project Plan: The Puget Sound Assessment and Monitoring Program Sediment Monitoring Component. Washington State Department of Ecology. Olympia, WA

Hart Crowser. 2007. Control of Toxic Chemicals in Puget Sound. Phase 1: Initial estimate of loadings. Washington Department of Ecology. Olympia, WA

Hickie, B.E., Ross, P.E., MacDonald, R.W., and Ford, J.K.B. 2007. Killer whales (*Orcinus orca*) face protracted health risks associated with lifetime exposure to PCBs. Environmental Science and Technology 41(18): 6613-6619.

Incardona, J.P., Carls, M.G., Day, H.L., Sloan, C.A., Bolton, J.L., Collier, T.K., and Scholz, N.L. 2009. Cardiac Arrhythmia Is the Primary Response of Embryonic Pacific Herring (Clupea pallasi) Exposed to Crude Oil during Weathering. Environmental Science & Technology 43(1): 201-207.

Johnson, L., Collier, T.K., and Stein, J.E. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conserv: Mar. Freshw. Ecosyst. 12: 517-538.

Johnson, L.L., Lomax, D.P., Myers, M.S., Olson, O.P., Sol, S.Y., O'Neill, S.M., West, J., and Collier, T.K. 2008. Xenoestrogen exposure and effects in English sole (*Parophrys vetulus*) from Puget Sound, WA. Aquatic Toxicology 88(1): 29-38.

Kimbrough, K.L., Lauenstein, G.G., Christensen, J.D., and Apeti, D.A. 2008. An assessment of two decades of contaminant monitoring in the Nation’s Coastal Zone. National Oceanic and Atmospheric Administration/National Center for Ocean Science Technical Memorandum, Silver Spring, MD.

Krahn, M.M., Hanson, M.B., Baird, R.W., Boyer, R.H., Burrows, D.G., Emmons, C.K., Ford, J.K.B., Jones, L.L., Noren, D.P., Ross, P.S., Schorr, G.S., and Collier, T.K. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54(12): 1903-1911.

Krahn, M.M., Wade, P.R., Kalinowski, S.T., Dalheim, M.E., Taylor, B.L., Hanson, M.B., Ylitalo, G.M., Angliss, R.P., Stein, J.E., and Waples, R.S. 2002. Status Review of Southern Resident Killer Whales (*Orcinus orca*) under the Endangered Species Act. Edited by U.S.D.o. Commerce. p. 133.

Krahn, M.M., Myers, M.S., Burrows, D.G., and Malins, D.C. 1984. Determination of metabolites of xenobiotics in the bile of fish from polluted waterways. Xenobiotica 14(8): 633-646.

Larsson, P., Andersson, A., Broman, D., Nordbäck, J., and Lundberg, E. 2000. Persistent organic pollutants (POPs) in pelagic systems. Ambio 29(4-5): 202-209.

Long, E.R., Dutch, M., Aasen, S., Welch, K., and Hameedi, M.J. 2005. Spatial extent of degraded sediment quality in Puget Sound (Washington State, U.S.A.) based upon measures of the sediment quality triad. Environmental Monitoring and Assessment 111: 173-222.

Long, E.R., Dutch, M., Aasen, S., Welch, K., and Hameedi, M.J. 2003. Chemical contamination, acute toxicity in laboratory tests, and benthic impacts in sediments of Puget Sound: a summary of results of the joint 1997-1999 Ecology/NOAA survey. Publication Number 03-03-049, or Technical Memorandum NOS NCCOS CCMA 163, Washington Department of Ecology, Lacey, WA.

Meador, J. 2006. Rationale and Procedures for Using the Tissue-Residue Approach for Toxicity Assessment and Determination of Tissue, Water, and Sediment Quality Guidelines for Aquatic

Organisms. Human and Ecological Risk Assessment: An International Journal 12(6): 1018 - 1073.

Meador, J.P. 2002. Use of tissue and sediment-based threshold concentrations of polychlorinated biphenyls (PCBs) to protect juvenile salmonids listed under the US Endangered Species Act. Aquatic Conservation: Marine and Freshwater Ecosystems 12: 493-516.

Muir, D.C.G., and Howard, P.H. 2006. Are There Other Persistent Organic Pollutants? A Challenge for Environmental Chemists. Environmental Science & Technology 40(23): 7157-7166.

Myers, M.S., Anulacion, B.F., French, B.L., Reichert, W.L., Laetz, C.A., Buzitis, J., Olson, O.P., Sol, S., and Collier, T.K. 2008. Improved flatfish health following remediation of a PAH-contaminated site in Eagle Harbor, Washington. Aquatic Toxicology 88: 277-288.

Myers, M.S., Landahl, J.T., Krahn, M.M., Johnson, L.L., and McCain, B.B. 1990. Overview of studies on liver carcinogenesis in English sole from Puget Sound; Evidence for a xenobiotic chemical etiology I: Pathology and epizootiology. Science of the Total Environment 94: 33-50.

Myers, M.S., Landahl, J.T., Krahn, M.M., and McCain, B.B. 1991. Relationships between hepatic neoplasms and related lesions and exposure to toxic chemicals in marine fish from the U.S. West Coastal. Environmental Health Perspectives 90: 7-15.

O'Connor, T.P. 2002. National distribution of chemical concentrations in mussels and oysters in the USA. Marine Environmental Research 53(2): 117-143.

O'Neill, S.M., and West, J.E. 2009. Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138(3): 616-632.

Partridge, V., Weakland, S., Long, E., Welch, K., Dutch, M., and Jones, M. 2009. Urban Waters Initiative, 2007; Sediment Quality in Elliott Bay. Washington Department of Ecology. Olympia, WA

Peterson, C.H., Rice, S.D., Short, J.W., Esler, D., Bodkin, J.L., Ballachey, B.E., and Irons, D.B. 2003. Long-Term Ecosystem Response to the Exxon Valdez Oil Spill. Science 302(5653): 2082-2086.

Puget Sound Action Team. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Seattle, WA

Puget Sound Estuary Program. 1987. Recommended protocols for fish patholgy studies in Puget Sound. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by Tetra Tech, Inc., Bellevue, WA for U.S. Environmental Protection Agency, Region 10, Seattle, WA. (Looseleaf).

Puget Sound Estuary Program. 1988. Elliott Bay Action Program: Analysis of Toxic Problem Areas prepared for US Environmental Protection Agency, Region X, Seattle, WA.

Puget Sound Estuary Program. 1989a (revised). Recommended guidelines for measuring organic compounds in Puget Sound sediment and tissue samples. In Recommended Protocols and Guidlines for Measuring Selected Environmental Variables in Puget Sound. Prepared by PTI Environmental Services, Bellevue, WA. for U.S. Environmental Protection Agency, Region 10, Seattle, WA (Looseleaf).

Puget Sound Estuary Program. 1989b (Revised). Recommeded guidelines for measuring inorganic compounds in Puget Sound sediment and tissue samples. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by PTI Environmental Services, Bellevue, WA. (Looseleaf) for U.S. Environmental Protection Agency, Region 10, Seattle, WA.

Puget Sound Estuary Program. 1990. Recommended guidelines for sampling demersal fish. In Recommended Protocols and Guidelines for Measuring Selected Environmental Variables in Puget Sound. Prepaired by PTI Environmental Services, Bellevue, WA. for

U.S. Environmental Protection Agency, Region 10, Seattle, WA (Looseleaf).

Puget Sound Estuary Program. 1996a. Recommended guidelines for measuring metals in Puget Sound marine water, sediment and tissue samples, Prepared for the U.S. Environmental Protection Agency, Region 10, Seattle, WA, and Puget Sound Water Quality Authority by King County Environmental Lab, Seattle, WA, Seattle, WA. (Looseleaf).

Puget Sound Estuary Program. 1996b. Recommended guidelines for measuring organic compounds in Puget Sound marine water, sediment and tissue samples, Prepared for the U.S. Environmental Protection Agency, Region 10, Seattle, WA, and Puget Sound Water Quality Authority by King County Environmental Lab, Seattle, WA, Seattle, WA. (Looseleaf).

Ross, P.S., Ellis, G.M., Ikonomou, M.G., Barrett-Lennard, L.G., and Addison, R.F. 2000. High PCB concentrations in free-ranging Pacific killer whales, *Orcinus orca*: effects of age, sex and dietary preference. Marine Pollution Bulletin 40: 504-515.

Ross, P.S., Jeffries, S.J., M.B.Yunker, Addison, R.F., Ikonomou, M.G., and Calambokidis, J.C. 2004. Harbor seals (Phoca vitulina) in British Columbia, Canada, and Washington State, USA reveal a combination of local and global polychlorinated biphenyl, dioxin, and furan signals. Environmental Toxicology and Chemistry 23(1): 157-165.

Ross, P.S. 2006. Fireproof killer whales (*Orcinus orca*): flame retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Science 63: 224-234.

Shen, G.T. 1995. Panel Findings and Recommendations: Based on the Final Comprehensive Review of the Puget Sound Ambient Monitoring Program, Puget Sound Water Quality Authority, P.O. Box 40900, Olympia, WA 98504-0900, Olympia, WA.

Sloan, J., and Gries, T. 2008. Dioxins, furans, and other contaminants in surface sediment and English sole collected from greater Elliott Bay (Seattle). Washington Department of Ecology. Olympia, WA

Spromberg, J.A., John, B.M., and Landis, W.G. 1998. Metapopulation dynamics: Indirect effects and multiple distinct outcomes in ecological risk assessment. Environmental Toxicology and Chemistry 17(8): 1640-1649.

Vines, C.A., Robbins, T., Griffin, F.J., and Cherr, G.N. 2000. The effects of diffusible creosote-derived compounds on development in Pacific herring (*Clupea pallasi*). Aquatic Toxicology 51(2): 225-239.

Weisbrod, A.V., Burkhard, L.P., Arnot, J., Mekenyan, O., Howard, P.H., Russom, C., Boethling, R., Sakuratani, Y., Traas, T., Bridges, T., Lutz, C., Bonnell, M., Woodburn, K., and Parkerton, T. 2007. Workgroup Report: Review of Fish Bioaccumulation Databases Used to Identify Persistent, Bioaccumulative, Toxic Substances. Environmental Health Perspectives 115(2): 255-261.

West, J.E., O'Neill, S.M., Lippert, G.R., and Quinnell, S.R. 2001. Toxic contaminants in marine and anadromous fish from Puget Sound, Washington: Results from the Puget Sound Ambient Monitoring Program Fish Component, 1989-1999. Technical Report. FTP01-14, Washington Department of Fish and Wildlife, Olympia, WA.

West, J.E., O'Neill, S.M., and Ylitalo, G.M. 2008. Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (*Clupea pallasi*) populations in the Puget Sound (USA) and Strait of Georgia (Canada). Science of the Total Environment 394(2-3): 369-378.

Wolfe, M.F., Schwartz, G.J.B., Singaram, S., Mielbrecht, E.E., Tjeerdema, R.S., and Sowby, M.L. 2001. Influence of dispersants on the bioavailability and trophic transfer of petroleum hydrocarbons to larval topsmelt (*Atherinops affinis*). Aquatic Toxicology 52(1): 49-60.

Harmful Algal Blooms

Background

Rapid growth and accumulation of phytoplankton or other algae can cause algal blooms. Bloom-forming algae that have harmful effects on people or wildlife are commonly termed harmful algal blooms (HABs). In Puget Sound, HABs may be caused by phytoplankton such as dinoflagellates of the genus *Alexandrium*, diatoms of the genus *Psuedo-nitzchia*, raphidophytes of the genus *Heterosigma* or by ulvoid seaweeds. Suspension-feeding bivalves, such as mussels, clams and oysters, can accumulate biotoxins to dangerous levels during HAB events, leading to illness such as paralytic shellfish poisoning (PSP) or amnesic shellfish poisoning (ASP) when the shellfish are ingested by humans, marine mammals and marine birds (Nishitani and Chew 1984, Hallegraeff 1993).

The Washington Department of Health(WDOH) Office of Shellfish Safety and Water Protection regularly monitors biotoxin levels in both recreational and commercial shellfish areas in Puget Sound. The Washington State Public Health Laboratory supports the WDOH through the analysis of shellfish samples. When high levels are detected in sample tissues, shellfish harvest areas are closed in order to protect shellfish consumers from biotoxin-related illness. Closures can have significant effects on commercial, recreational, and subsistence harvest. Episodes of high biotoxin levels are currently unpredictable in time or space due to the interaction of multiple poorly understood environmental factors (Moore et al. 2009).

Paralytic Shellfish Poisoning

Seasonal restrictions on commercial and recreational shellfish harvest due to PSP, sometimes known as "red tide", are common in Washington. The biotoxin that causes PSP temporarily interferes with the transmission of nerve impulses in warm-blooded animals. Symptoms of PSP in humans range from nausea, vomiting, numbness of the lips and tongue and muscle paralysis to death by cardio-respiratory arrest. There is no known antidote for the toxin, and cooking does not destroy the toxin.

Several microscopic organisms that naturally exist in marine water produce the PSP toxin. The species that causes PSP in Washington marine waters is the dinoflagellate *Alexandrium catenella* (Determan et al. 2001). Alexandrium is typically present in small numbers; however, when environmental conditions are favorable, rapid reproduction and accumulation can occur, and shellfish can accumulate the toxin to dangerous levels during such bloom events (Zingone and Enevoldsen 2000, Moore et al. 2009).

WDOH closes areas for shellfish harvest when PSP toxin levels equal or exceed the Food and Drug Administration standard of 80 micrograms (μg) of toxin per 100 grams of shellfish tissue. Areas are not reopened until testing has confirmed that the PSP toxin has declined to a safe level. Butter clams may experience extended closures because they typically retain the PSP toxin longer than other shellfish (up to one year).

Sentinel Mussel Monitoring Program

The Sentinel Mussel Monitoring Program is an early warning system for marine biotoxins established by WDOH. Mussels generally register PSP toxin levels more quickly than other shellfish. Consequently, mussels are used as "sentinels" to determine whether PSP toxins are increasing in a given area. Under this monitoring program, mussels are placed in cages and set in strategic growing areas throughout Puget Sound. Mussel samples are then collected either biweekly or monthly and tested for levels of PSP. Rising PSP levels in these mussels trigger more targeted and frequent sampling regimens in other shellfish species in the affected area.

With assistance from local health jurisdictions, local tribes, the Puget Sound Restoration Fund, and volunteers, WDOH maintained and monitored 69 collection sites in 2008 (WDOH 2009). In addition to the sentinel mussel locations, commercial mussels were routinely monitored at Westcott Bay in San Juan Island and at Penn Cove in Whidbey Island.

Amnesic Shellfish Poisoning

Domoic acid is a naturally-occurring toxin produced by a species of microscopic marine diatoms of the genus *Pseudo-nitzschia*. The human illness known as ASP or domoic acid poisoning is caused by eating fish, shellfish or crab containing the toxin. ASP can result in gastrointestinal and neurological disorders within 24-48 hours of toxic shellfish consumption by humans, and can be life-threatening. There is no antidote for domoic acid poisoning and cooking does not destroy the toxin.

The razor clam and Dungeness crab fisheries on the outer coast of Washington State have incurred losses due to occurrences of domoic acid over the past two decades. In the fall of 1991, domoic acid was first detected in razor clams off the coast of Washington and caused several mild cases of ASP (Horner and Postel 1993). This prompted WDOH to begin monitoring all major shellfish growing areas for domoic acid. Research shows that razor clams accumulate domoic acid in the edible tissue (foot, siphon, and mantle) and are slow to rid themselves of the toxin (Wekell et al. 1994) due to the presence of a high affinity glutamate binding protein (Trainer and Bill 2004). However, razor clams can continue to function in marine environments with high concentrations of domoic acid (Trainer and Bill 2004), resulting in extended closures of shellfish beds of the outer coast of Washington. In Dungeness crab, domoic acid primarily accumulates in the viscera. The level of domoic acid determined to be unsafe for human consumption is 20 parts per million (ppm) in molluscan shellfish and 30 ppm for Dungeness crab viscera. Dungeness crab harvest areas are closed when three out of six individual crab viscera equals or exceeds 30 ppm.

Within Puget Sound, the first occurrence of domoic acid was in blue mussels harvested in Kilisut Harbor in 2003 (Bill et al. 2006), raising concerns about the possibility of shellfish closures similar to those on the outer coast. Puget Sound was presumed to be less susceptible to domoic acid closures due to the absence of harvested species (razor clams and Dungeness crab) that retain domoic acid for long periods. Many shellfish species that are harvested in Puget Sound, such as mussels, littleneck clams, and oysters, are able to depurate domoic acid over a period of hours or days (Novaczek 1992), whereas the ability of other species such as geoduck to retain or release domoic acid has not yet been determined (Trainer et al. 2007).

Heterosigma

While not responsible for illnesses in humans, blooms of the small, unicellular, flagellated raphidophyte *Heterosigma akashiwo* have been shown to kill fish through the likely production of neurotoxins that disrupt respiratory and osmoregulatory gill functions (Khan et al. 1997, Hard et al. 2000). Farmed fish are particularly susceptible to mortality from increased concentrations of *Heterosigma* (Chang et al. 1990, Hard et al. 2000). Increased water column stratification and high temperatures have both been correlated with *Heterosigma* blooms although the precise causes for blooms remain uncertain (Bearon et al. 2006, O'Halloran et al. 2006).

Ulvoids

Blooms of ulvoid seaweeds are manifested by large quantities of green algal biomass washing up on beaches where decomposition occurs or in seagrass beds where mortality of seagrass through smothering is possible (den Hartog 1994, Nelson and Lee 2001). The thin blade-like morphology of ulvoids is thought to contribute to their ability to respond quickly to favorable environmental conditions such as increased nutrients and light (e.g., Littler and Littler 1980). As such, they have can competitively displace other algal species and seagrasses (e.g., den Hartog 1994, Anderson et al. 1996, Valiela et al. 1997). While not typically associated with the production of toxins, there is emerging evidence that ulvoid algae can produce allelopathic compounds that are detrimental to the development and growth of invertebrate larvae and other algae (Nelson et al. 2003a, Van Alstyne et al. 2007). Two genera of ulvoid seaweeds are common in Puget Sound: *Ulva* (which includes the former genus *Enteromorpha*) and *Ulvaria* (formerly referred to as *Monostroma*)(Nelson et al. 2003b). Despite their morphological similarity, these genera differ ecologically. A combination of field and lab observations conducted by Nelson and colleagues has demonstrated that *Ulva* is more tolerant of desiccation stress, produces lower levels of allelopathic compounds and is found more commonly in intertidal habitats whereas *Ulvaria* is less tolerant of desiccation stress, produces higher levels of allelopathic compounds and is more commonly found in subtidal habitats (Nelson et al. 2003a, Nelson et al. 2003b, Nelson et al. 2008, Nelson et al. 2010).

Status

PSP and ASP

In 2008, only 12 of 2,798 samples (0.4%) of shellfish tested by the Washington State Public Health Laboratory detected levels of PSP toxins greater than 1,000 micrograms and no PSP-related illnesses in humans were reported (WDOH 2009) (Table 1). However, 23 subtidal <u>geoduck clamtracts</u> were closed due to elevated PSP toxin levels and two general closures for "all shellfish species" occurred. Notably, one <u>geoduck</u> tract closure included a recall of 3,368 lbs of <u>geoduck clams</u>. In 2008, the highest PSP levels in blue mussels were found in Mystery Bay, Kilisut Harbor (Table 1).

Table 1. Areas of highest PSP levels in 2008 (WDOH 2009)

Date	Harvest Area	Species	Toxin Level*

08/10/2008	Mystery Bay, Kilisut Harbor	Blue Mussel	2,602
06/17/2008	Semiahmoo Marina, Drayton Harbor	Blue Mussel	1,831
08/11/2008	Scow Bay, Kilisut Harbor	Blue Mussel	1,779
09/25/2008	Dockton, Quartermaster Harbor	Blue Mussel	1,462
06/17/2008	Birch Bay Village, Birch Bay	Blue Mussel	1,456
08/06/2008	Fort Flagler, Kilisut Harbor	Blue Mussel	1,347
11/12/2008	Ediz Hook, East Straits	Blue Mussel	1,097

- micrograms per 100 grams of shellfish meat tissue

Approximately 12 Dungeness crab and 1,318 molluscan shellfish samples were tested by WDOH for domoic acid in 2008. The low sample size for crabs was driven by lack of toxin in the first 12 samples, which prompted a halt in further testing of Dungeness crab. There were no shellfish closures due to high levels of domoic acid in 2008, nor any reported ASP illnesses (WDOH 2009). The highest levels of domoic acid observed in Puget Sound molluscs in 2008 were at Squaxin Passage and Budd Inlet (Table 2).

Table 2. Areas of highest domoic acid levels in 2008 (WDOH 2009)

Date	Harvest Area	Species	Toxin Level*
06/24/2008	Squaxin Passage	Blue Mussel	3
06/19/2008	Budd Inlet	Blue Mussel	3
01/07/2008	Kalaloch Beach North	Razor Clam	3
11/06/2008	Long Beach Reserve	Razor Clam	2
10/01/2008	Sequim Bay	Blue Mussel	2
06/24/2008	South Tacoma Narrows	Blue Mussel	2

- parts per million per 1 gram of shellfish meat tissue

Heterosigma

Heterosigma has been reported in various locations in Puget Sound and has been linked to fish mortality at fish farms in Puget Sound (Hershberger et al. 1997, Tyrrell et al. 2002). Despite the potential problem of financial damage to fish farms from Heterosigma or of mortality of wild fish from *Heterosigma* blooms, available data the spatial variation of both *Heterosigma* occurrence and the frequency of associated fish mortality events is limited.

Ulvoids

A study conducted by Nelson et al. (2003b) assessing biomass of ulvoids in locations in Puget Sound (Figure 1) in summer of 2000 found that the species composition, depth, and abundance of ulvoids was variable throughout Puget Sound (Figure 2). In a more detailed analysis linking ulvoid biomass to abiotic variables on the coast of Blakely Island in the San Juan Archipelago, Nelson et al. (2003b) found that increased biomass was positively correlated with increased nitrogen, a finding that is consistent with studies conducted in other locations (e.g., Sfriso et al. 1992, Anderson et al. 1996).

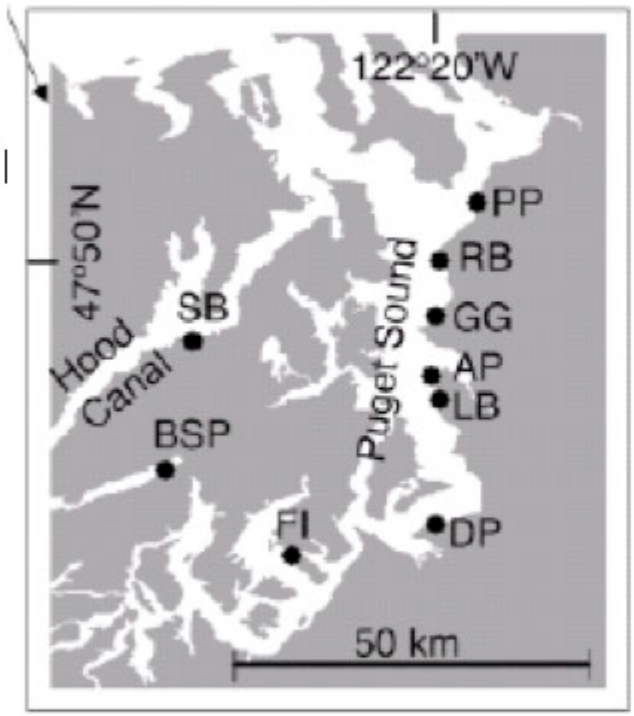


Figure 1. Sampling locations for ulvoid algae conducted by Nelson et al.(2003b)(Reprinted with permission from Botanica Marina and De Gruyter Publishing).

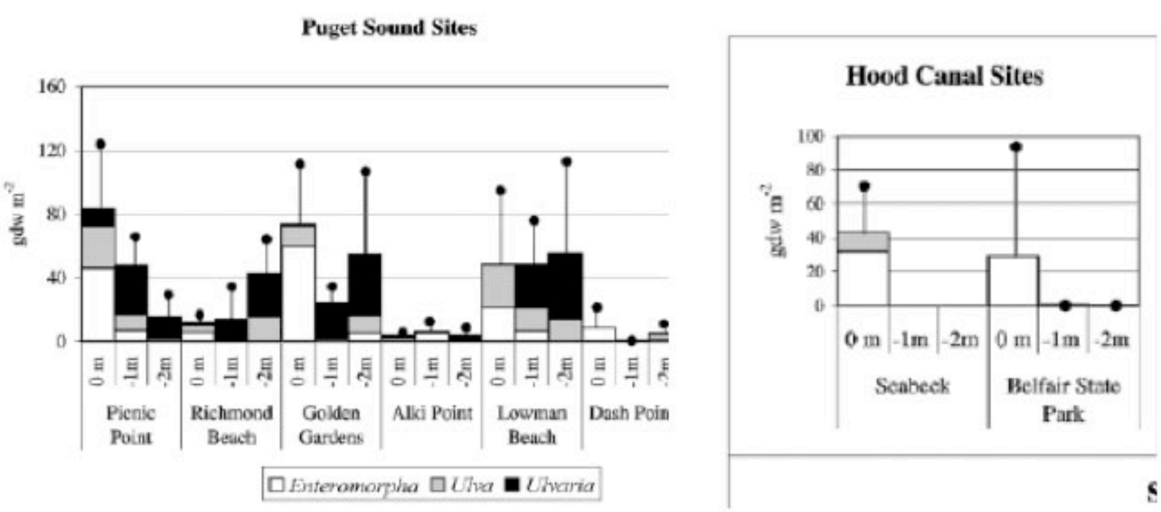


Figure 2. Biomass (mean + 1SD) by genus and by depth if ulvoid algae at locations throughout Puget Sound (Nelson et al. 2003b)(Reprinted with permission from Botanica Marina and De Gruyter Publishing).

Trends

PSP

Harmful algal blooms of *Alexandrium* were widespread and prevalent in the northern regions of Puget Sound (e.g., Sequim and Discovery Bays) in the 1950s and 1960s, but extended southward in the 1970s and 1980s to inner regions of Puget Sound (Trainer et al. 2003). More recent occurrences of PSP toxins in Washington shellfish and crab have been variable. Although high levels of PSP were detected in many years between 1990 and 2006, in some years (e.g., 1995, 2007, 2008) PSP toxin levels remained low (Figure 3). Despite this variability, the frequency of instances of high levels of PSP toxins detected by WDOH monitoring in Washington State has increased since 1957(Figure 3), a trend that is consistent with a worldwide increase in PSP toxic events since the 1950s (Nishitani and Chew 1984, Hallegraeff 1993, Trainer et al. 2003, Maso and Garces 2006).

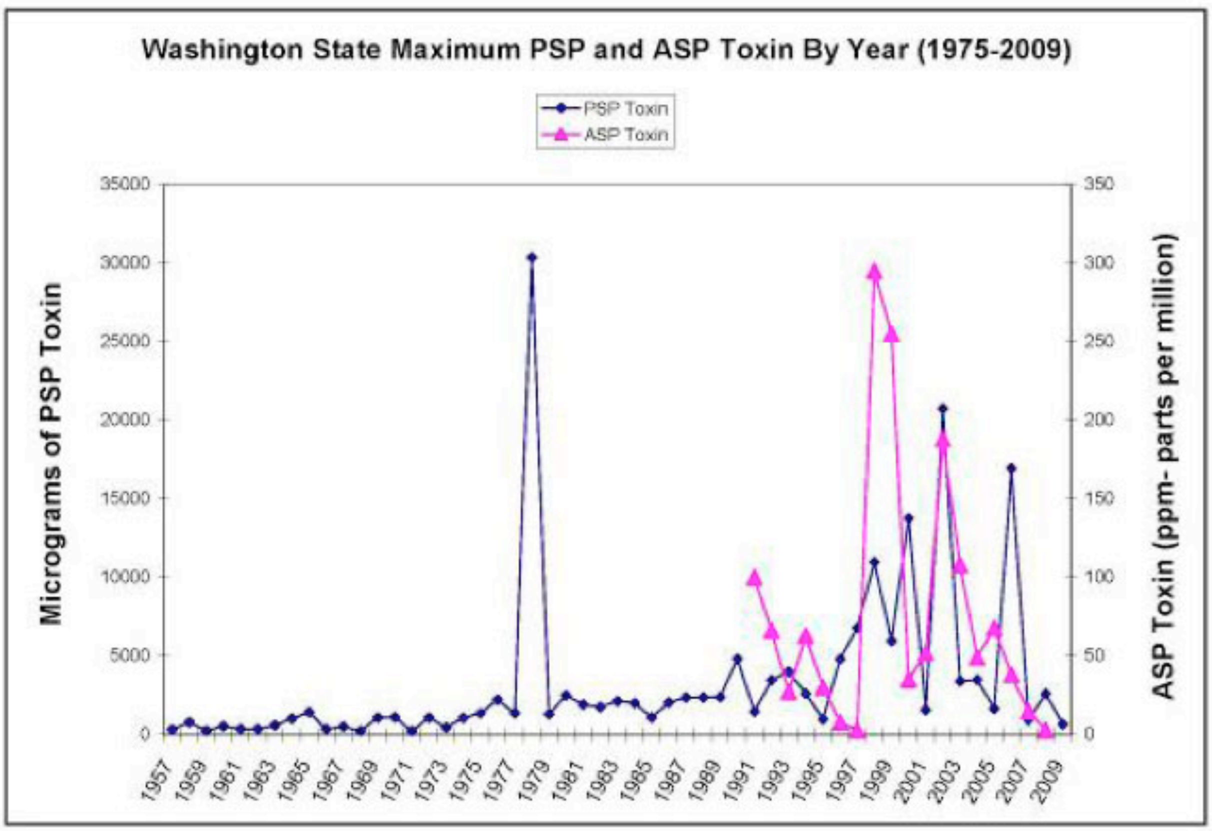


Figure 3. Annual maximum concentrations of PSP and ASP toxins observed in Washington State (WDOH, reprinted from Puget Sound Partnership 2009).

Moore et al. (2009) caution the use of PSP levels in shellfish tissues as a proxy for algal cell density in the water column due to the difference in accumulation and depuration rates of shellfish species. To investigate trends and possible relationships with large-scale climate and local environmental factors, Moore et al. (2009) analyzed PSP levels in blue mussels between 1993 and 2007. A combination of warm air and water temperatures and low streamflow appears to be favorable for PSP toxin accumulation in mussels, but advanced warning of events may be constrained by the same factors as for weather prediction, and is therefore limited to approximately one to two weeks (Moore et al. 2008, Moore et al. 2009). No increase in the frequency, magnitude, duration, or geographic scope of HAB events was detected, yet a significant basin-wide trend for closures to be imposed earlier in the year was observed over the period.

ASP

Blooms of *Pseudo-nitzchia* continue to affect Washington's outer coast since the first fisheries closure due to ASP toxins in 1991. Exceptional years of domoic acid-associated beach, razor clam, and Dungeness crab closures in Washington include 1991, 1998-1999, 2002-2003, and 2005 (Horner and Postel 1993, Trainer et al. 2007)(Figure 3). The prolonged closures of 1998-1999 and 2002-2003 (>1.5 years) resulted in significant commercial, recreational, and tribal shellfish harvest losses in Washington State (Dyson and Huppert in press, corrected proof). Out of concern for ASP toxins in the highly populated Puget Sound region, WDOH has monitored throughout Puget Sound since 1991. *Pseudo-nitzschia* blooms were reported in Puget Sound in 2003 and 2005, causing concern that blooms could impact the valuable fisheries there (Trainer et al. 2009).

Heterosigma

The bloom-forming alga *Heterosigma akashiwo* is recognized as a potential problem in Puget Sound. Despite a number of current studies on this HAB-forming alga, data are not yet available to determine spatial and temporal trends in *Heterosigma* abundances or the frequency of toxic events in Puget Sound.

Ulvoids

Published accounts of temporal trends in ulvoid abundances in Puget Sound are lacking. At least one investigation currently is underway to estimate ulvoid abundance from archival video surveys.

Uncertainties

Trend analysis of harmful algal blooms is difficult due to the lack of understanding about the dynamics that drive them, although this is an area of active research (e.g., Bearon et al. 2006, Nelson et al. 2008, Moore et al. 2009). Environmental conditions such as circulation, temperature, sunlight, nutrients, and salinity as well as the presence of algal predators, parasites and algal disease organisms all likely play a role in the formation, magnitude, and persistence of blooms. While PSP and ASP toxin levels currently are monitored and reported by WDOH, published data regarding spatial and temporal trends in *Heterosigma* and ulvoid abundances in Puget Sound are lacking.

Summary

Harmful algal blooms in Puget Sound have been variable over the past two decades, but appear to be increasing since WDOH began monitoring in 1957. Current monitoring efforts are not sufficient to provide accurate forecasting of ASP and PSP-related bloom events beyond one to two weeks, but forecasting could be improved by increased temporal and spatial scale and automated devices. While there is emerging concern about blooms of *Heterosigma* and ulvoids, , data that address these concerns currently is lacking for Puget Sound.

Literature Cited

Anderson, R. J., P. M. S. Monteiro, and G. J. Levitt. 1996. The effect of localised eutrophication on competition between *Ulva lactuca* (Ulvaceae, Chlorophyta) and a commercial resource of *Gracilaria verrucosa* (Gracilariaceae, Rhodophyta). Hydrobiologia 327:291-296.

Bearon, R. N., D. Grunbaum, and R. A. Cattolico. 2006. Effects of salinity structure on swimming behavior and harmful algal bloom formation in *Heterosigma akashiwo*, a toxic raphidophyte. Marine Ecology-Progress Series 306:153-163.

Bill, B. D., F. H. Cox, R. A. Horner, J. A. Borchert, and V. L. Trainer. 2006. The first closure of shellfish harvesting due to domoic acid in Puget Sound, Washington, USA. African Journal of Marine Science 28:435-440.

Chang, F. H., C. Anderson, and N. C. Boustead. 1990. 1st Record of a *Heterosigma* (Raphidophyceae) bloom with associated mortality of cage-reared salmon in Big Glory Bay, New Zealand New Zealand Journal of Marine and Freshwater Research 24:461-469.

den Hartog, C. 1994. Suffocation of a littoral *Zostera* bed by *Enteromorpha radiata*. Aquatic Botany 47:21-28.

Determan, T. A., S. Washington . Office of Food, P. Shellfish, and P. Puget Sound Ambient Monitoring. 2001. Paralytic shellfish poisoning (PSP) patterns in Puget Sound shellfish, year 2000 : a report for the Puget Sound Ambient Monitoring Program. Office of Food Safety and Shellfish Programs, Washington State Dept. of Health, [Olympia, Wash.].

Dyson, K., and D. D. Huppert. in press, corrected proof. Regional economic impacts of razor clam beach closures due to harmful algal blooms (HABs) on the Pacific coast of Washington. Harmful Algae In Press, Corrected Proof.

Hallegraeff, G. 1993. A review of harmful algae blooms and their apparent global increase. Phycologia 32:79 - 99.

Hard, J. J., L. Connell, W. K. Hershberger, and L. W. Harrell. 2000. Genetic variation in mortality of chinook salmon during a bloom of the marine alga *Heterosigma akaskiwo*. Journal of Fish Biology 56:1387-1397.

Hershberger, P. K., J. E. Rensel, A. L. Matter, and F. B. Taub. 1997. Vertical distribution of the chloromonad flagellate *Heterosigma carterae* in columns: implications for bloom development. Canadian Journal of Fisheries and Aquatic Sciences 54:2228-2234.

Horner, R. A., and J. R. Postel. 1993. Toxic diatoms in western Washington waters (US West coast). Hydrobiologia 269:197-205.

Khan, S., O. Arakawa, and Y. Onoue. 1997. Neurotoxins in a toxic red tide of *Heterosigma akashiwo* (Raphidophyceae) in Kagoshima Bay, Japan. Aquaculture Research 28:9-14.

Littler, M. M., and D. S. Littler. 1980. The evolution of thallus form and survival strategies in benthic marine macroalgae: Field and laboratory tests of a functional form model. The American Naturalist 116:25.

Maso, M., and E. Garces. 2006. Harmful microalgae blooms (HAB); problematic and conditions that induce them. Marine Pollution Bulletin 53:620-630.

Moore, S., V. Trainer, N. Mantua, M. Parker, E. Laws, L. Backer, and L. Fleming. 2008. Impacts of climate variability and future climate change on harmful algal blooms and human health. Environmental Health 7:S4.

Moore, S. K., N. J. Mantua, B. M. Hickey, and V. L. Trainer. 2009. Recent trends in paralytic shellfish toxins in Puget Sound, relationships to climate, and capacity for prediction of toxic events. Harmful Algae 8:463-477.

Nelson, T. A., K. Haberlin, A. V. Nelson, H. Ribarich, R. Hotchkiss, K. L. V. Alstyne, L. Buckingham, D. J. Simunds, and K. Fredrickson. 2008. Ecological and physiological controls of species composition in green macroalgal blooms. Ecology 89:1287-1298.

Nelson, T. A., and A. Lee. 2001. A manipulative experiment demonstrates that blooms of the macroalga *Ulvaria obscura* can reduce eelgrass shoot density. Aquatic Botany 71:149-154.

Nelson, T. A., D. J. Lee, and B. C. Smith. 2003a. Are "green tides" harmful algal blooms? Toxic properties of water-soluble extracts from two bloom-forming macroalgae, *Ulva fenestrata* and *Ulvaria obscura* (Ulvophyceae). Journal of Phycology 39:874-879.

Nelson, T. A., A. V. Nelson, and M. Tjoelker. 2003b. Seasonal and spatial patterns of "Green tides" (Ulvoid algal blooms) and related water quality parameters in the coastal waters of Washington state, USA. Botanica Marina 46:263-275. De Gruyter (Publisher).

Nelson, T. A., J. Olson, L. Imhoff, and A. V. Nelson. 2010. Aerial exposure and desiccation tolerances are correlated to species composition in "green tides" of the Salish Sea (northeastern Pacific). Botanica Marina 53:103-111.

Nishitani, L., and K. K. Chew. 1984. Recent developments in paralytic shellfish poisoning research. Aquaculture 39:317-329.

Novaczek, I., Madhyastha, M.S., Ablett, R.F., Donald, A., Johnson, G., Nijjar, M.S., Sims, D.E. 1992. Depuration of domoic acid from live blue mussels (*Mytilus edulis*). Can. J. Fish. Aquat. Sci. 49:312-318.

O'Halloran, C., M. W. Silver, T. R. Holman, and C. A. Scholin. 2006. *Heterosigma akashiwo* in central California waters. Harmful Algae 5:124-132.

Puget Sound Partnership. 2009. Ecosystem Status & Trends. Puget Sound Partnership, Seattle, WA.

Sfriso, A., B. Pavoni, A. Marcomini, and A. Orio. 1992. Macroalgae, nutrient cycles, and pollutants in the Lagoon of Venice. Estuaries and Coasts 15:517-528.

Trainer, V., B.-T. Eberhart, J. Wekell, N. Adams, L. Hanson, F. Cox, and J. Dowell. 2003. Paralytic shellfish toxins in Puget Sound, Washington State. J Shellfish Res 22:213 - 223.

Trainer, V. L., and B. D. Bill. 2004. Characterization of a domoic acid binding site from Pacific razor clam. Aquatic Toxicology 69:125-132.

Trainer, V. L., W. P. Cochlan, A. Erickson, B. D. Bill, F. H. Cox, J. A. Borchert, and K. A. Lefebvre. 2007. Recent domoic acid closures of shellfish harvest areas in Washington State inland waterways. Harmful Algae 6:449-459.

Tyrrell, J. V., L. B. Connell, and C. A. Scholin. 2002. Monitoring for *Heterosigma akashiwo* using a sandwich hybridization assay. Harmful Algae 1:205-214.

Valiela, I., J. McClelland, J. Hauxwell, P. J. Behr, D. Hersh, and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: Controls and ecophysiological and ecosystem cnsequences. Limnology and Oceanography 42:1105-1118.

Van Alstyne, K., L. Koellermeier, and T. Nelson. 2007. Spatial variation in dimethylsulfoniopropionate (DMSP) production in *Ulva lactuca* (Chlorophyta) from the Northeast Pacific. Marine Biology 150:1127-1135.

WDOH, W. D. o. H. 2009. 2008 Annual Report: Commercial and Recreational Shellfish Areas in Washington State. Office of Shellfish and Water Protection.

Wekell, J. C., J. R. Gauglitz, E.J.Barnett, H.J.Hatfield, C.L.Simons, and D. Ayres. 1994. Occurrence of domoic acid in Washington state razor clams (*Siliqua patula*) during 1991–1993. Natural Toxins 2:197-205.

Zingone, A., and H. O. Enevoldsen. 2000. The diversity of harmful algal blooms: a challenge for science and management. Ocean & Coastal Management 43:725-748.

Dissolved Oxygen (Hypoxia)

Background

Hypoxia, defined as dissolved oxygen (DO) concentrations less than 2 mg / L, has become widespread throughout estuaries and semi-enclosed seas throughout the world (Diaz 2001). While hypoxia may be permanent or intermittent, it is most commonly manifested as a seasonal disturbance, appearing in mid- to late summer after vertical stratification prevents replenishment of deep water DO. The duration, extent and magnitude of seasonal hypoxia has dramatically increased over the past few decades in response to anthropogenic eutrophication (Diaz and Rosenberg 2008) and is now a common and regular feature in marine ecosystems that have strong vertical stratification and low flushing rates. Additionally, climate change may be altering the frequency and intensity of hypoxic conditions in coastal ecosystems (Chan et al. 2008).

Hypoxia is an important concern because low dissolved oxygen can have direct and indirect effects on marine communities and natural resources. Hypoxia and anoxia can be lethal to animals when oxygen levels are depleted beyond species physiological tolerances. For sessile organisms who have limited capacity to seek out refuges from hypoxia, direct lethal impacts may be most severe (Diaz and Rosenberg 1995). Mobile species generally act to minimize exposure to low DO through distributional shifts to refuges that have higher DO levels. While these responses minimize direct lethal impacts of low DO, they can induce indirect ecological effects such as reduced feeding rates, enhanced vulnerability to predators and reduced growth rates (Breitburg 1992, Breitburg et al. 1997, Eby and Crowder 2002, Bell et al. 2003, Craig and Crowder 2005, Aumann et al. 2006).

Status of hypoxia in Puget Sound

In many regions of Puget Sound, low DO is a natural consequence of its deep fjord-like bathymetry, where the water column stratification and slow flushing rates lead to long residence times of deep water that is not in contact with the atmosphere. However, there is some evidence that DO levels were generally higher in the mid 20th century than they are today (Newton et al. 1995). This conclusion is based on a comparison of historical water quality sampling data to contemporary data that used comparable techniques (Figure 1). Changes in the intensity of low DO conditions over a time period of increased human activity suggests a role for anthropogenic activity in dictating hypoxia.

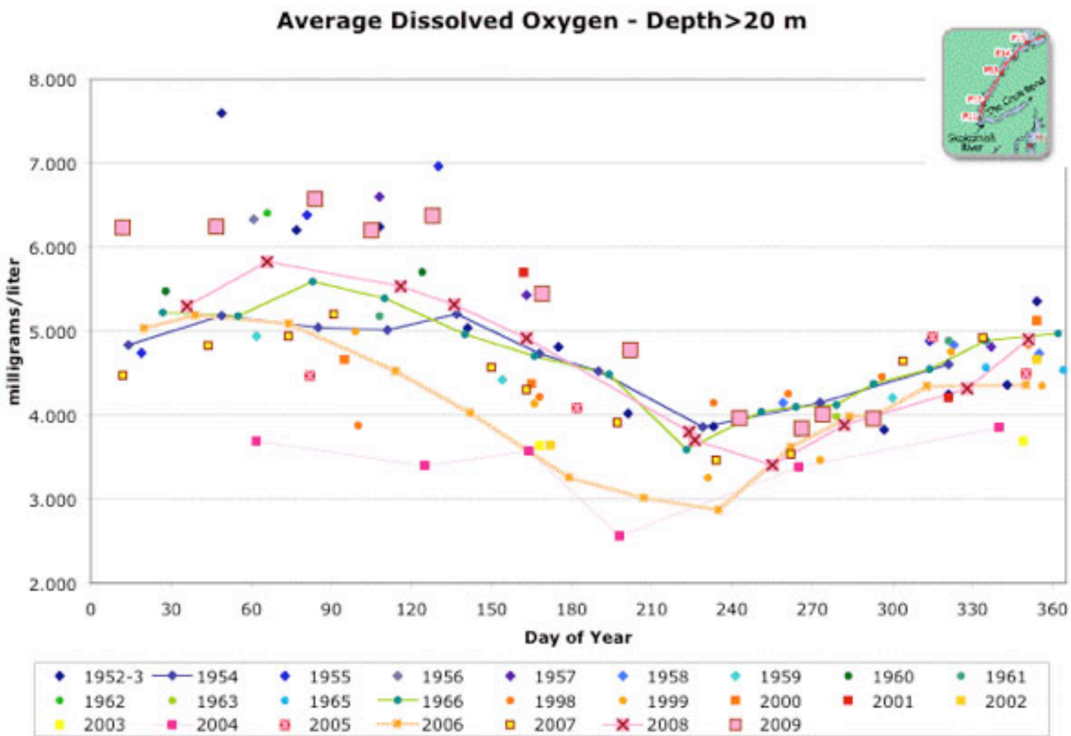


Figure 1. Integrated sub-surface water DO vs. day by sampling year in southern Hood Canal. Recent years with very low DO conditions (e.g., 2004, 2006) have no historical precedent over the period of record (1952 -1966). Data and analysis from Hood Canal Dissolved Oxygen Program: http://www.hoodcanal.washington.edu/. Figure produced by and used with permission from M.J. Warner, University of Washington.

Low dissolved oxygen is present seasonally in Puget Sound at several locations (Figure 2). Much of the southern one-half of Hood Canal now regularly experiences hypoxic conditions in mid- to late summer. Several regions within the south basin of Puget Sound are also prone to hypoxia (Albertson et al. 2007), especially Budd, Carr and Case Inlet (Albertson et al. 2002). Saratoga passage also is susceptible to low DO (Figure 2)(Albertson et al. 2002).

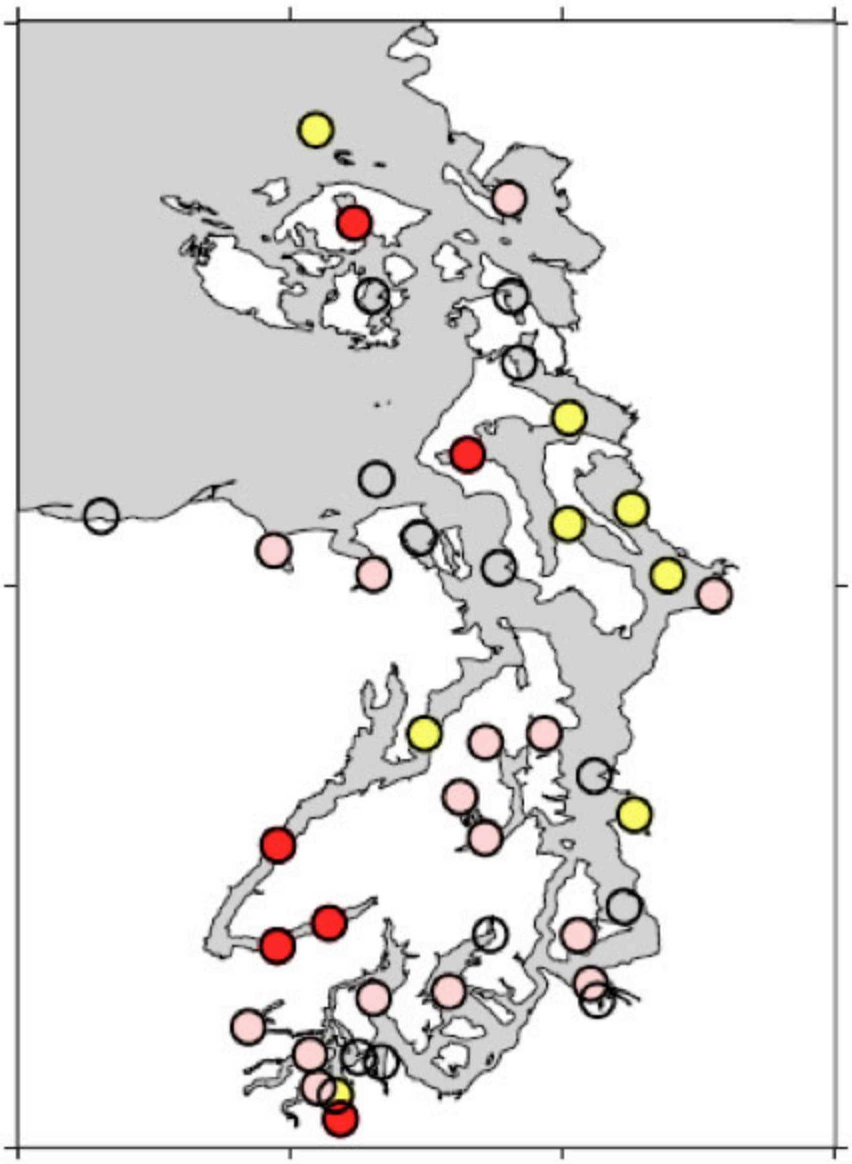


Figure 2. Washington State Department of Ecology water quality monitoring stations showing low (red, < 3 mg /l), stressful (yellow, <5 mg /l) and high (empty circles) DO, 1990-1997. Pink circles indicate stations likely to be to have low DO based on physico-chemical characteristics. Reprinted from Albertson et al. (2002) with permission from Washington Department of Ecology.

Since the mid-2000's, there has been a proliferation of monitoring efforts and web-based distribution of data, especially for Hood Canal. These include (1) monitoring via citizens that provides weekly along 6 stations that transverse Hood Canal (2) deploying of remove ocean observing systems (Oceanic Remote Chemical-optical Analyzer, ORCA) that provide high frequency monitoring of water conditions (3) routine monitoring via WA Department of Ecology.

These data can be downloaded from http://www.hoodcanal.washington.edu/. The expansion of data collection capacity has revealed the importance of oceanographic processes for determining the spatial patterning and temporal persistence of low DO in Hood Canal (Figure 3). In both Hood Canal and South Puget Sound, research activities are presently underway to develop high resolution models to predict DO levels and their sensitivity to surface flows and oceanographic conditions (Albertson et al. 2007). The below summary emphasizes insights gleaned from the study of Hood Canal, only because of the greater concentration of research activity in this basin.

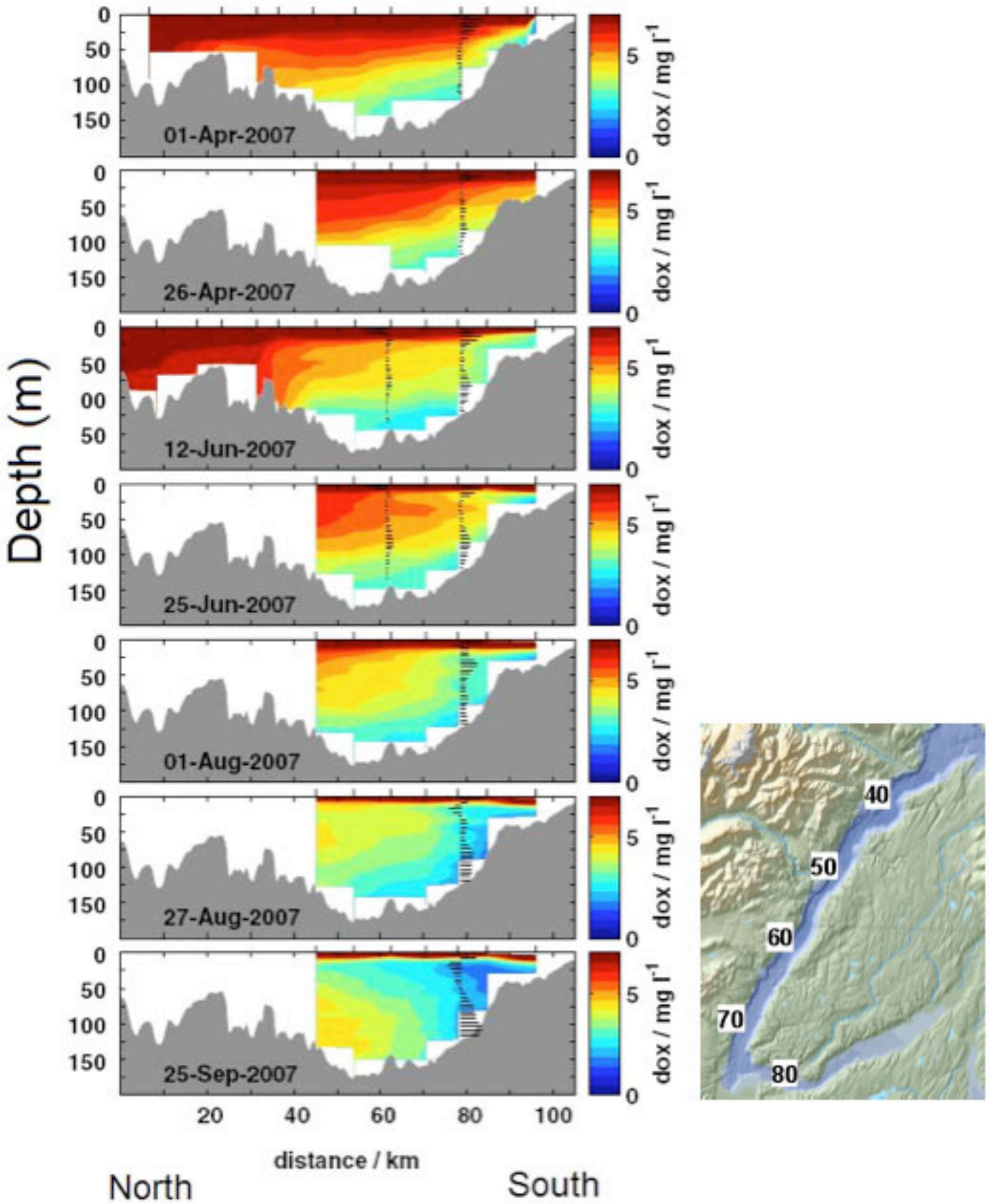


Figure 3. Cross sectional profiles of dissolved oxygen (DO) in Hood Canal, April – September 2007. Hypoxia first emerges at depth in the southern end of Hood Canal, and extends northward along the bottom until mid summer. In mid summer, the horizontal extent of hypoxia constricts southward but expands vertically through late summer / early autumn. Black bars indicate mean water velocities and direction. In the proposed study, the southern impact region spans roughly kilometers 70 – 80, while the north unimpacted region extends from 40 – 50. Inset map shows location of cross section distance markers (kilometers). Figure produced by Mickett and M.J. Warner and used with permission from M.J. Warner, University of Washington. http://www.hoodcanal.washington.edu/observations/ccross.jsp.

Anthropogenic Influences

Hypoxia is a symptom of eutrophication whereby excessive primary production fuels high rates of microbial respiration as sinking organic matter is decomposed in deep waters. Cultural eutrophication is caused by anthropogenic loading of nutrients that limited phytoplankton growth; in Puget Sound, dissolved inorganic nitrogen (DIN) is the primary limiting nutrient for primary producers (Newton et al. 1998). Thus, human activities that increase DIN loading directly promote hypoxic conditions. DIN enters the Puget Sound through multiple sources: (1) via the surrounding watershed via surface flow, groundwater, wastewater, and shallow septic systems (2) from recycling of nutrient from the sediments into the water column; (3) directly from the atmosphere and (4) from water exchange with the marine environment. Human activity primarily affects watershed-based inputs, although climate change could alter delivery of nitrogen from coastal marine waters.

Three primary anthropogenic activities are thought to be important in changing low DO conditions via DIN inputs into Puget Sound. The first is through the conversion of riparian vegetation from a community dominated by firs and cedars to one replaced with red alders (Busse and Gunkel 2001). As alders host symbiotic microorganisms that have the capacity to fix atmospheric nitrogen into a biologically available form, their current abundance may lead to increased nitrogen loading. The second is through shallow shoreline septic systems. A mass balance estimation of DIN loads to Hood Canal revealed that shallow ground water flow from shoreline septic systems contributed less than 5% of the total dissolved inorganic nitrogen to the upper water layer (Paulson et al. 2007). The third is from wastewater treatment plant discharge. In South Puget Sound, wastewater treatment comprises roughly one-half of the watershed-derived DIN loading (Roberts et al. 2008), but this component may be larger if water exchange with the central Basin is considered (Figure 4)(Roberts et al. 2008). At present, there are no published reports or papers that definitively implicate any single source as the most important cause of reduced DO.

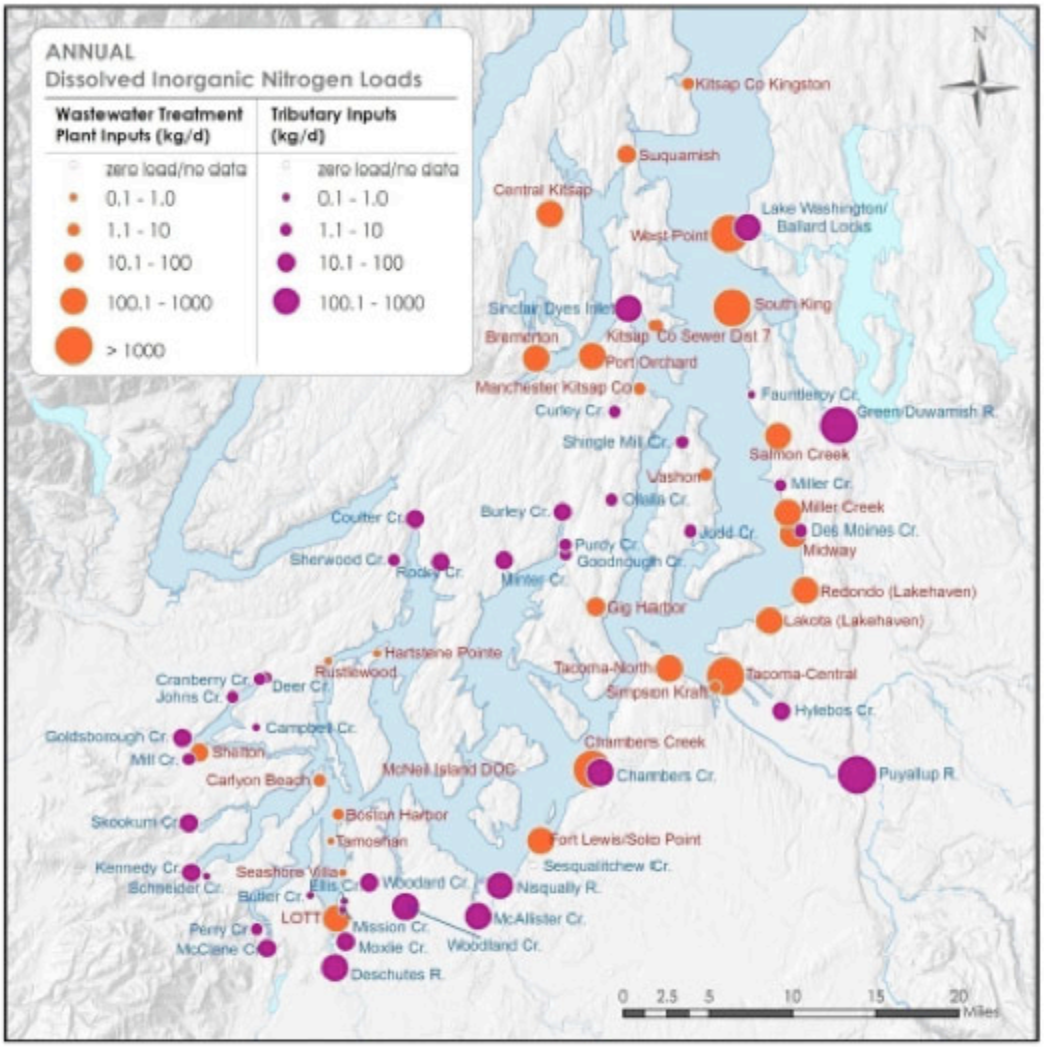


Figure 4. Annual DIN loads from freshwater surface flows and wastewater treatment plants. Reprinted from Roberts et al. (2008) with permission from Washington Department of Ecology.

Impacts on Biota

Hypoxia has been implicated in shifts in benthic infauna and in the pelagic community. Benthic infauna provide the most important source of food for most of the groundfish in Puget Sound so changes is these communities may have important ecological consequences. Long et al. (2007) demonstrated substantial shifts in community structure associated with water column dissolved oxygen levels below 3 mg/L. In general, the overall density of benthic infauna and species richness were reduced as dissolved oxygen decreased. Valero et al (2008) compared population dynamics of geoduck clams in southern and northern reaches of Hood Canal and implicated hypoxia as a significant factor in population declines in the southern region. Parker-Stetter and Horne (2009) described shifts in the distribution of pelagic organisms during a period of pronounced midwater anoxic zone during 2006, suggesting that severe midwater minimum layers can create a predation refuge for zooplankton. However, in the following year midwater oxygen minimum layers did not appear to affect the vertical distribution of fish and invertebrates, although it did appear to impact the rate of diel migration (Parker-Stetter et al. 2009). Palsson et al. (2008) described substantial vertical distributional shifts of rocky-reef associated fish species in response to the low dissolved oxygen event, but also noted that responses varied by species.

Several fish kill events in southern Hood Canal have been documented (2002, 2003, 2006), all occurring in late summer. Fish kill events correspond with a rapid vertical displacement of hypoxic / anoxic water, such that mobile fishes are unable to mount behavioral responses quickly enough to avoid exposure. The 2003 and 2006 fish kill events were differentiated by the primary species affect: copper rockfish were the dominant species affected in the 2003 event, while lingcod were more impacted by the 2006 event (Palsson et al. 2008). The ratios of dead : total observed copper rockfish ranged from 0 – 26%, while for lingcod these rations ranged from 3 – 37% (Palsson et al. 2008)

CONTENT PENDING REVIEW

Added: 10/7/2010

Author: Dr. Tim Essington, School of Aquatic and Fisheries Science, University of Washington

Essington and Paulsen (2010) used a comparative approach to ask whether there was evidence of hypoxia on densities of demersal fish and macroinvertebrates in southern Hood Canal. They found strong evidence supporting the hypothesis that sessile macroinvertebrate densities are strongly reduced by hypoxia: the five main species sampled were generally reduced in abundance by ca. 90% compared to what would be expected based from the reference sites. In contrast, there was little evidence for persistent density reductions in mobile fauna. However, mobile macroinvertebrates and fishes exhibited significant density reductions in southern Hood Canal during late summer when hypoxia was present, presumably due to behavioral distributional responses that displaced individuals from southern Hood Canal. The large reduction in demersal species' densities suggests substantial effects of hypoxia in Hood Canal even at oxygen levels that were marginally hypoxic (2 mg / l). They conclude that understanding the full ecological consequence of hypoxia will require a greater knowledge on the spatial extent of distributional shifts and their effects on competitive and predator–prey interactions.

Reference:

Essington, TE and Paulsen, CE 2010. Quantifying hypoxia impacts on an estuarine demersal community using a hierarchical ensemble approach. Ecosystems. 13: published on-line prior to print doi:10.1007/s10021-010-9372-z

Uncertainties

Identifying the ultimate causes of hypoxia and policy responses that might mitigate them remains a high priority. Because of high interannual variability, it is not possible to discern whether the intensity or spatial extent of hypoxia has been growing over recent years. Moreover, the long-term effects of regular exposure to seasonal hypoxia on communities and food webs has not yet been published. Valuable species such as geoduck clams and Dungeness crabs may be adversely affected by hypoxic conditions, though it is not yet possible to definitively quantify the contribution of hypoxia to putative population declines in hypoxia-impacted regions.

Literature Cited

Albertson, S. L., J. Bos, G. Pelletier, and M. Roberts. 2007. Estuarine flow int he south basin of Puget Sound and its effects on near-bottom dissolved oxygen. WA Department of Ecology, Olympia, WA.

Albertson, S. L., K. Erickson, J. A. Newton, G. Pelletier, R. A. Reynolds, and M. Roberts. 2002. South Puget Sound Water Quality Study Phase 1. WA Department of Ecology, Olympia, WA.

Aumann, C. A., L. A. Eby, and W. F. Fagan. 2006. How transient patches affect population dynamics: The case of hypoxia and blue crabs. Ecological Monographs 76:415-438.

Bell, G. W., D. B. Eggleston, and T. G. Wolcott. 2003. Behavioral responses of free-ranging blue crabs to episodic hypoxia. II. Feeding. Marine Ecology-Progress Series 259:227-235.

Breitburg, D. 1992. Episodic hypoxia in Chesapeake Bay: Interacting effects of recruitment, behavior, and physical disturbance. Ecological Monographs 62:525-546.

Breitburg, D. L., T. Loher, C. A. Pacey, and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. Ecological Monographs 67:489-507.

Busse, L. B. and G. Gunkel. 2001. Riparian alder fens - Source or sink for nutrients and dissolved organic carbon?:1. Effects of water level fluctuations. Limnologica 31:307-315.

Chan, F., J. A. Barth, J. Lubchenco, A. Kirinich, H. Weeks, W. T. Peterson, and B. A. Menge. 2008. Emergence of anoxia in the California current large marine ecosystem. Science 319:920.

Craig, J. K. and L. B. Crowder. 2005. Hypoxia-induced habitat shifts and energetic consequences in Atlantic croaker and brown shrimp on the Gulf of Mexico shelf. Marine Ecology-Progress Series 294:79-94.

Diaz, R. J. 2001. Overview of hypoxia around the world. Journal of Environmental Quality 30:275-281.

Diaz, R. J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. : Oceangraphy and Marine Biology : an Annual Review 33:245-303.

Diaz, R. J. and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321:926-929.

Eby, L. A. and L. B. Crowder. 2002. Hypoxia-based habitat compression in the Neuse River Estuary: context-dependent shifts in behavioral avoidance thresholds. Canadian Journal Of Fisheries And Aquatic Sciences 59:952-965.

Long, E. R., M. E. Dutch, S. Aasen, K. I. Welch, V. A. Partridge, and D. A. Shull. 2007. Relationships between the composition of the benthos and sediment and water quality parameters in Hood Canal: Task IV Hood Canal Dissolved Oxygen Program. WA Department of Ecology, Olympia, WA.

Newton, J. A., M. Edie, and J. Summers. 1998. Primary productivity in Budd Inlet during 1997: Seasonal patterns of variation and controlling factors.in Puget Sound Research 1998.

Newton, J. A., A. L. Thomson, L. B. Eisner, G. A. Hannach, and S. L. Albertson. 1995. Dissolved oxygen concentrations in Hood Canal: are conditions different than forty years ago?in Puget Sound Research '95.

Palsson, W. A., R. E. Pacunski, T. R. Parra, and Beam.J. 2008. The effects of hypoxia on marine fish populations in Southern Hood Canal, Washington. American Fisheries Society Symposium 64:255-280.

Parker-Stetter, J. L. and J. K. Horne. 2009. Nekton distribution and midwater hypoxia: a seasonal, diel prey refuge? Estuaries and Coasts 81:13-18.

Parker-Stetter, S. L., J. K. Horne, and M. M. Langness. 2009. The influence of midwater hypoxia on nekton vertical migration. ICES Journal of Marine Science 66:1296-1302.

Paulson, A. J., C. P. Konrad, L. M. Frans, M. Noble, C. Kendall, E. G. Josberger, R. L. Huffman, and T. D. Olsen. 2007. Freshwater and saline loads of dissolved inorganic nitrogen to Hood Cana and Lynch Cove, Washington. U.S. Gelogical Survey, Reston, VA.

Roberts, M., J. Bos, and S. L. Albertson. 2008. South Puget Sound dissolved oxygen study interim data report. WA Department of Ecology, Olympia, WA.

Valero, J. L., B. Lee, D. Armstrong, L. Orensanz, A. Parma, R. Hilborn, B. Sizemore, and T. Palzer. 2008. Population dynamics and historic trends of geoduck clams under episodic low dissolve oxygen conditions in Hood Canal. Journal Of Shellfish Research 27:462-463.

Eutrophication of Marine Waters

Background

Eutrophication of water bodies occurs when high levels of nutrients fuel high rates of primary production and accumulation of algal biomass, either as macroalgae or phytoplankton. Some ecosystems are naturally eutrophic, but in others human activity causes ecosystems to undergo transformations into a eutrophic state. This is termed cultural eutrophication, and is the primary concern in evaluating the status of marine waters of Puget Sound.

The primary cause of cultural eutrophication is human actions (e.g., land use, wastewater, agriculture) that increase the loadings of nutrients that limit algal growth (Carpenter et al. 1998). In Puget Sound (like many estuaries), dissolved inorganic nitrogen (DIN) is the primary limiting nutrient (Newton and Van Voorhis 2002). Research efforts have therefore focused on measuring the availability of DIN and on the rates of delivery from alternative sources. In general, DIN in Puget Sound can come from (1) the surrounding watershed via surface flow, groundwater, wastewater, and shallow septic systems; (2) recycling of nutrients from the sediments into the water column; (3) directly from the atmosphere; and (4) exchange with the coastal ocean. Human activity primarily affects watershed-based inputs, although climate change could alter delivery of nitrogen from coastal marine waters through its effects on coastal upwelling.

The vulnerability of an ecosystem to cultural eutrophication depends on several factors. Generally, strong vertical mixing can act to reduce the effects of nutrient enrichment via inducing light limitation on planktonic producers. Many areas of Puget Sound experience regular mixing through tidal exchange processes that could act to reduce the effects of anthropogenic DIN loading (Figure 1), but some are less well mixed and are therefore vulnerable to eutrophication. Such areas tend to be inlets with few freshwater inputs, and deep fjord-like basins that have limited exchange with surrounding waters (e.g., Hood Canal, South Puget Sound; Figure 1). A second major consideration is the extent to which primary production is already limited by DIN. This depends in large part on the availability of N from other sources: if DIN supply from other sources is relatively large, impacts of smaller additions of total N from anthropogenic sources may be relatively small. In Puget Sound, much of the DIN derives from exchange with coastal marine waters via exchanges in the Strait of Juan de Fuca and subsequently in the major sub-basins of Puget Sound (Mackas and Harrison 1997). A final consideration is the residence time of surface waters: if systems are rapidly flushed then surface waters containing anthropogenic DIN will be displaced quickly.

Figure 1. Sampling stations containing strong and persistent vertical stratification (red), based on WA Department of Ecology and PRISM data. Sites denoted by yellow and green are at lower risk of eutrophication. Reprinted from U.S. E.P.A. Region 10 Puget Sound Georgia Basin Ecosystem Indicators (for supporting references, see U.S. Environmental Protection Agency (2006).

In Puget Sound, the extent of DIN- limitation on algae varies strongly with space and time (Newton and Van Voorhis 2002) (Figure 2). In general, response of phytoplankton to nutrient enrichment is greatest during May – Aug. Nutrient responses in these months correspond to a drawing down of available DIN in the surface mixed layer during the spring, when phytoplankton production and standing stocks are the greatest (Newton and Van Voorhis 2002, Stark et al. 2008) (Figure 3a, 3b).

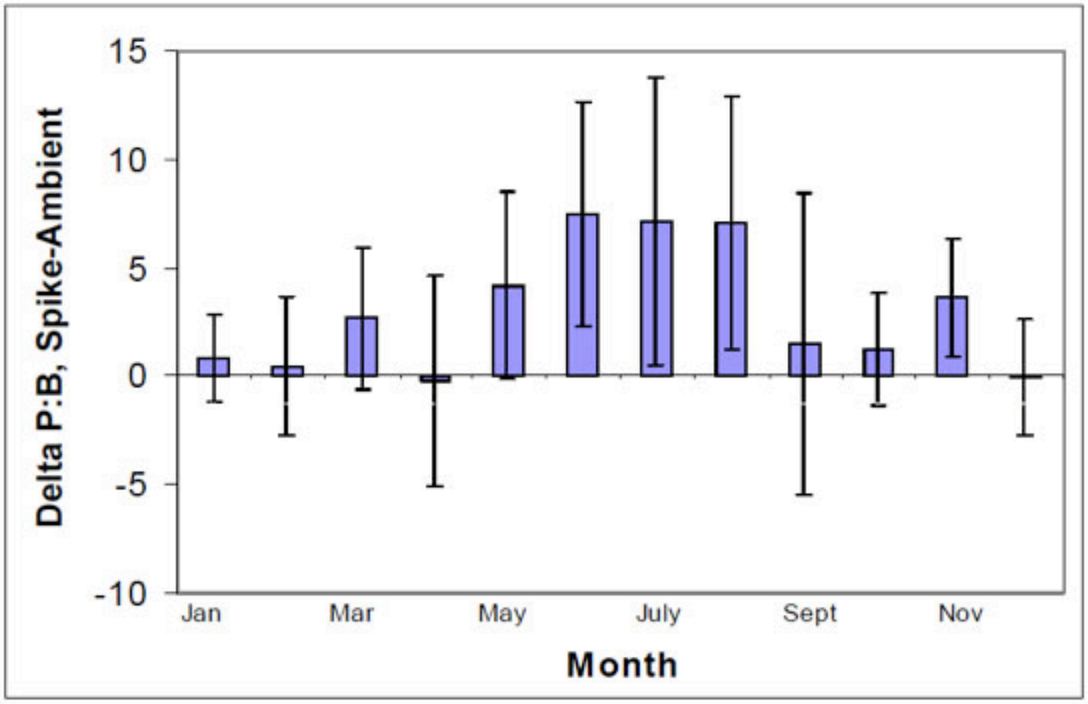


Figure 2. Change in phytoplankton production (production : biomass; PB) in response to nutrient spike. Bars represent averages taken over multiple sites. Nutrient limitation is greatest in May - August. Reprinted from Newton and Van Voorhis (2002) with permission from Washington Department of Ecology.

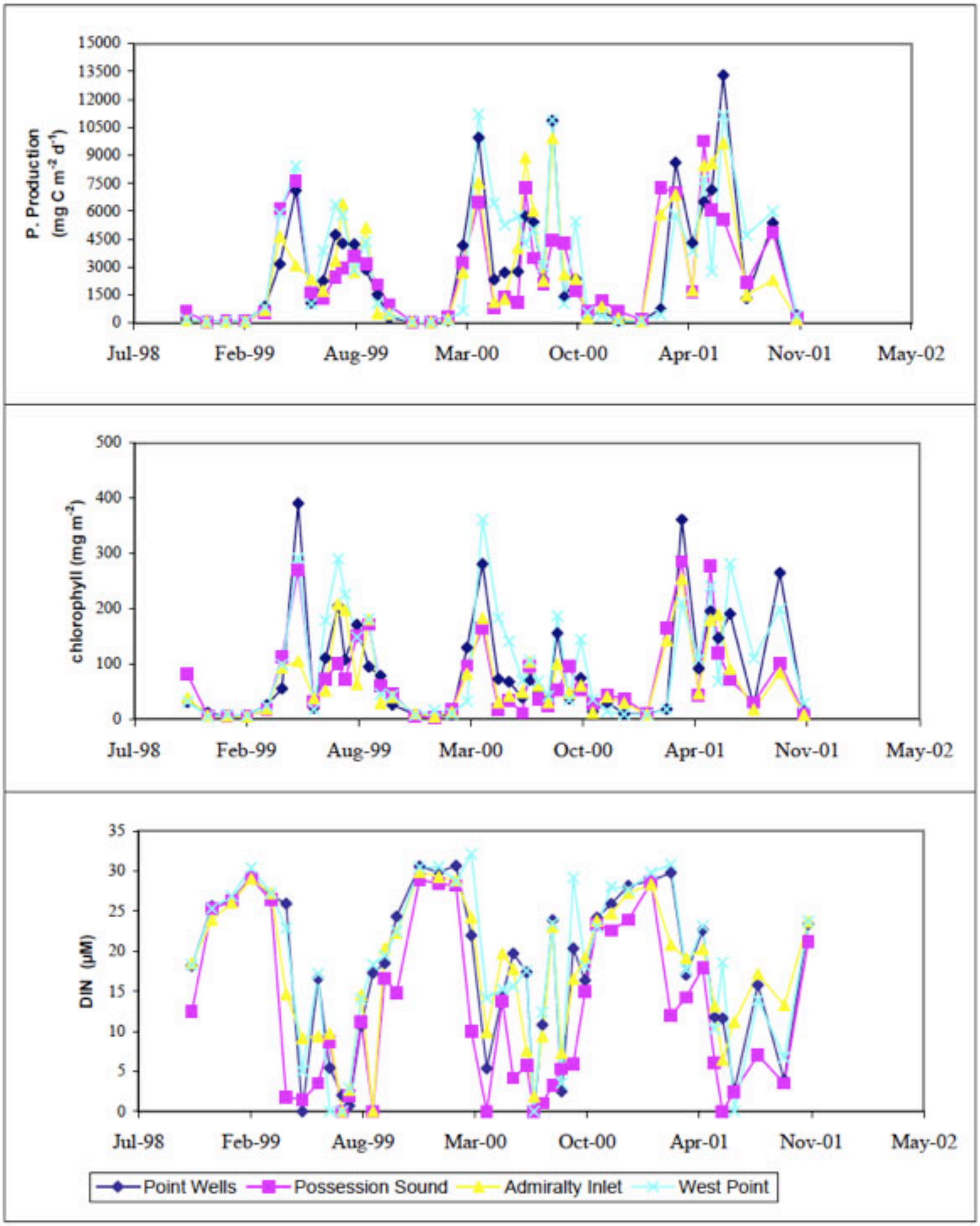


Figure 3a. Seasonal patterns of primary productivity, Chl.A and DIN at four sites, 1998 -2001. Reprinted from Newton and Van Voorhis (2002) with permission from Washington Department of Ecology.

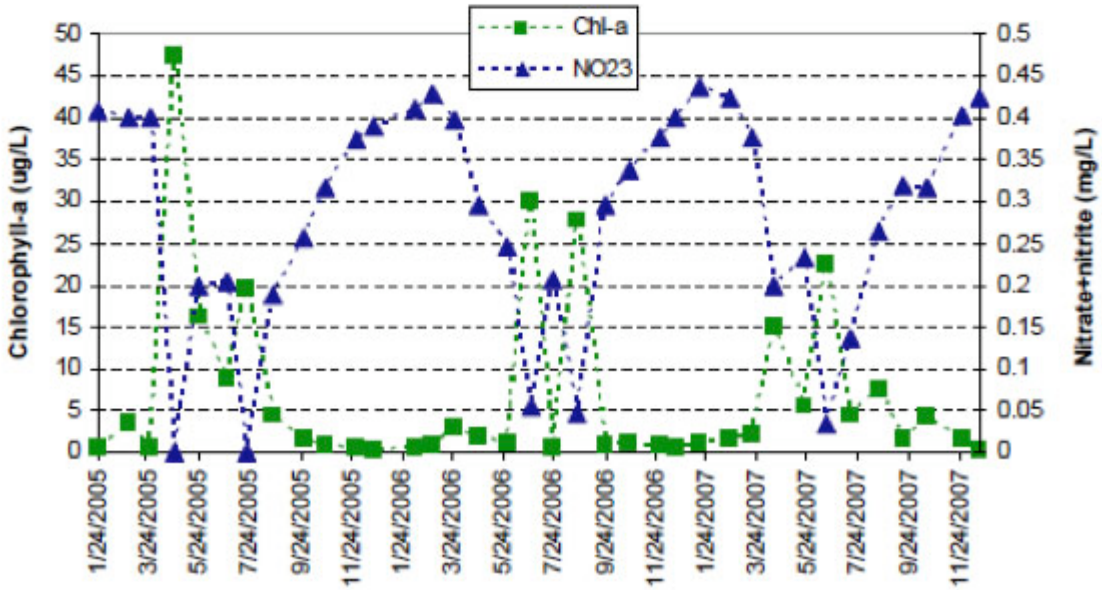


Figure 3b. Seasonal patterns of chlorophyll level and Nitrate/ Nitrite at Point Wells monitoring station, 2005-2007. Phytoplankton blooms are associated with a draw down of available DIN. Reprinted from Stark et al. (2008) with permission from King County Department of Natural Resources and Parks.

Monitoring Programs

Several entities conduct regular water quality monitoring within Puget Sound. The Washington State Department of Ecology conducts monthly sampling at several sites throughout Puget Sound (Figure 4). King County conducts monthly sampling at 14 offshore stations and 18 beach / nearshore stations in Central Puget Sound. The University of Washington PRISM program conducts biannual sampling at 39 stations throughout Puget Sound (Figure 5). The Hood Canal Dissolved Oxygen Program maintains 4 moorings that provide high-frequency monitoring of water quality conditions, and King County maintains three active moorings in central Puget Sound. Although the design of some of these monitoring programs have evolved over time to adapt to emerging issues, core sites have been maintained so that long-term trends can be evaluated(Newton et al. 2002). Detailed QA/ QC procedures for many of these programs are well documented (Washington State Department of Ecology 2006b, Albertson et al. 2007a).

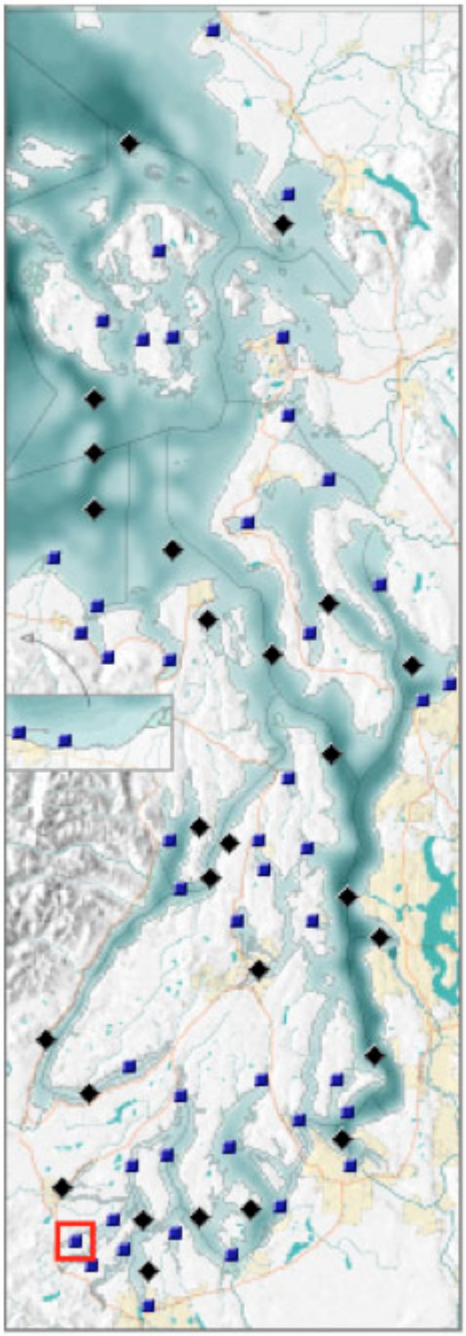


Figure 4. Location of Department of Ecology sampling sites. Used with permission from Department of Ecology.

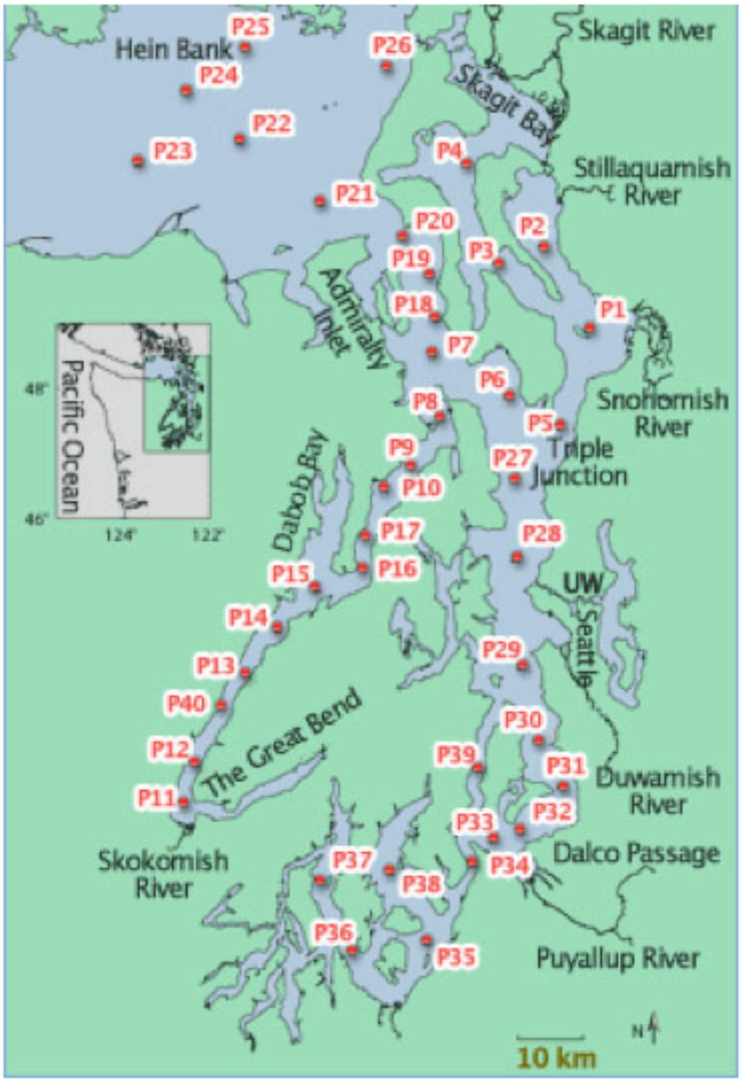


Figure 5. Location of PRISM sampling sites. Used with permission from University of Washington, publisher of web pages for Hood Canal Dissolved Oxygen Program.

Status and Trends

Several groups synthesize monitoring information to evaluate the status of eutrophic conditions throughout Puget Sound and in specific regions that are particularly vulnerable to eutrophication. King County uses a modified version of the Oregon Water Quality Index (Cude 2001) to combine information on dissolved oxygen, DIN and strength of vertical stratification to derive a single number that can be used to assess high to moderate eutrophication risk. In central Puget Sound, index values have been low since 2004 (the first year the index was calculated), except for 2007 when about 20% of the sampling sites showed moderate or high risk. We are unaware of any review process that evaluates the effectiveness of this modified index for predicting the onset of eutrophic conditions. The Department of Ecology published regular updates of their monitoring program up to 2002 (Newton et al. 2002) but no longer continues that reporting format. The Department of Ecology internet portal provides direct access to monitoring data and the results of a ranking algorithm by area for multiple water and sediment quality metrics (Washington State Department of Ecology 2006a).The most recent assessment year is 2008 and the 2010 assessment is scheduled to be complete by September 2010. Briefly, this index scores sample sites on a scale from 1 to 5. Scores of 1 to 3 indicate no water quality impairment, while scores of 4 and 5 indicated impairment. A score of 5 triggers action regarding Total Maximum Daily Loads. No synthetic analysis of the spatio-temporal extent of regions scoring 5 on this scale has been conducted, although the previous iteration of the Puget Sound Science Update reported DO levels at DOE monitoring stations that had very low (< 3 mg /l), low (2 mg /l = 5 mg / l) and high (>5 mg /l) DO levels. In a review of estuarine conditions nationwide, Bricker et al. (2007) reported moderate to high levels of eutrophication in several regions of Puget Sound and high risk for worsening conditions in Hood Canal and South Puget Sound (Table 1). These rankings are based on surveys rather than an explicit and consistent data analysis effort. Albertson et al. (2002) demonstrated eutrophication symptoms in several regions throughout south Puget Sound (Figure 6). Eutrophication in southern Hood Canal has been well documented (Newton 2007) (see Dissolved Oxygen).

Table 1. Summary of current status, future outlook and status of influencing factors by location, From Bricker et al. 2007. Status levels and risk are assigned based on surveys of local experts, not on quantitatively defined categories.

	Influencing Factors	Eutrophic Conditions	Future Outlook (risk of worsening conditions)
Central Puget Sound	Unknown	Moderate	Unknown
South Puget Sound	Unknown	Moderate	High
Skagit Bay / Whidbey Basin	Unknown	Moderate	Unknown
Hood Canal	High	High	High
Sequim / Discovery Bay	Unknown	Moderate / High	Unknown
Port Orchard Sound	Unknown	Moderate	Unknown

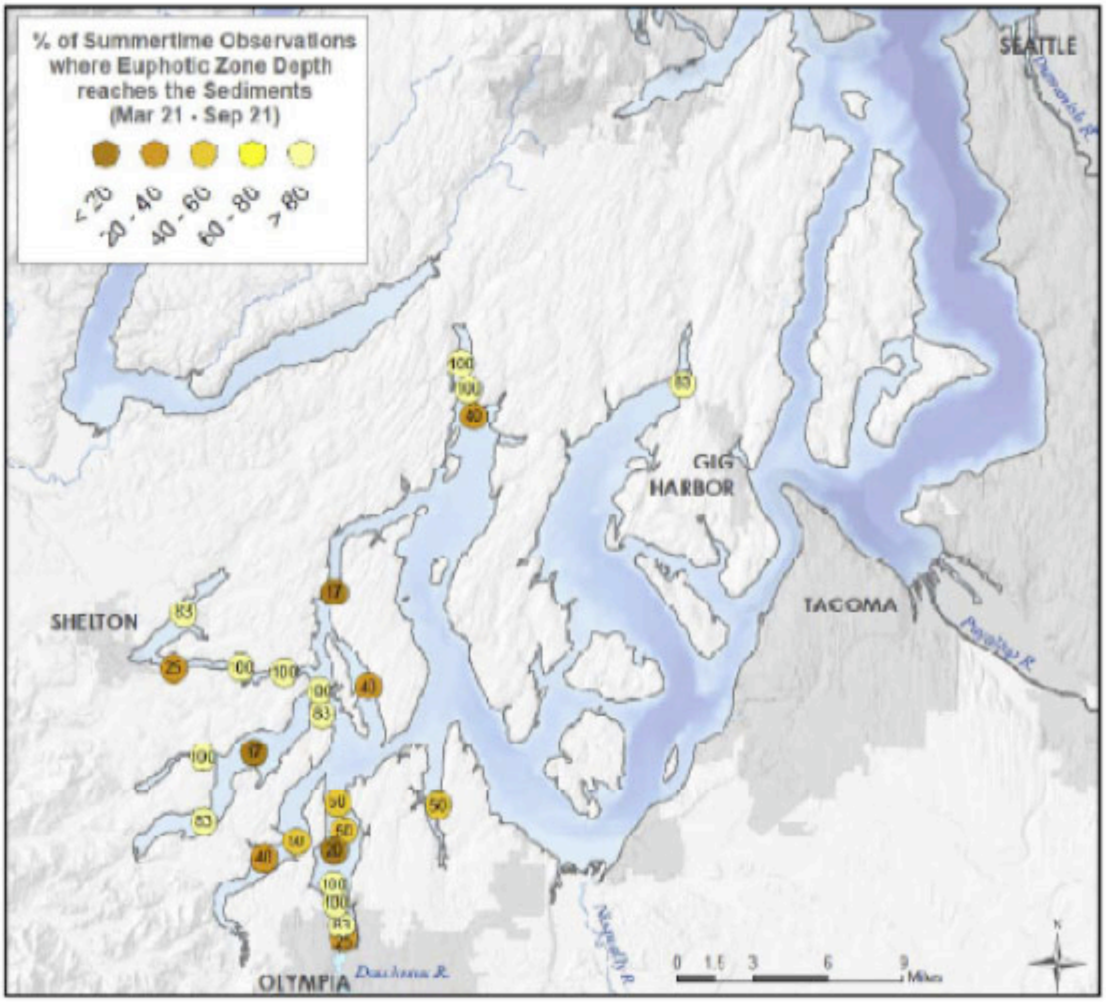


Figure 6. Summertime water clarity in South Puget Sound, 2006 – 2007. Dark points indicate sites with reduced frequencies of high water clarity. Reprinted from Albertson et al. (2002) with permission from Washington Department of Ecology.

Uncertainties

Ongoing research is working to develop detailed biophysical models of Puget Sound that will be useful for gauging the contributions of human activities to changes in trophic status of Puget Sound (Albertson et al. 2007b) and for identifying the most effective policy interventions to prevent worsening conditions. Our present understanding of the threats to Puget Sound is sufficient for identifying areas at risk of cultural eutrophication on the basis of stratification intensity and surface water residence time. We are aware that the Washington State Department of Ecology is presently developing a novel water quality index that may be effective in gauging the current water quality status throughout Puget Sound. Mapping this and other indices against the indicators used in NOAA's national assessment may permit comparisons across ecosystems to better gauge the status of Puget Sound. Future eutrophication status may be affected by climate change through its effects on coastal upwelling intensity, ambient air temperature and timing of freshwater flows.

Literature Cited

Albertson, S., J. Bos, K. Erickson, C. Maloy, G. Pelletier, and M. Roberts. 2007a. South Puget Sound Water Quality Study Phase 2: Dissolved Oxygen Quality Assurance Project Plan. Washington State Department of Ecology.

Albertson, S. L., J. Bos, G. Pelletier, and M. Roberts. 2007b. Estuarine flow in the south basin of Puget Sound and its effects on near-bottom dissolved oxygen. WA Department of Ecology, Olympia, WA.

Albertson, S. L., K. Erickson, J. A. Newton, G. Pelletier, R. A. Reynolds, and M. Roberts. 2002. South Puget Sound Water Quality Study Phase 1. WA Department of Ecology, Olympia, WA.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. Effects of nutrient enrichment in the nation's estuaries : a decade of change. National Centers for Coastal Ocean Science, Silver Spring MD.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559-568.

Cude, C. G. 2001. Oregon water qualit index: a tool for evaluating water quality management effectiveness. Journal fo the American Water Resources Association 37:125-137.

Mackas, D. L. and P. J. Harrison. 1997. Nitrogenous nutrient sources and sinks in the Juan de Fuca Strait / Strait of Georgia / Puget Sound Estuarine system: Assessing the potential for eutrophication. Estuarine, Coastal and Shelf Science 44:1-44.

Newton, J. 2007. Hood Canal: the complex factors causing low dissolved oxygen events require ongoing research, monitoring and modeling. Pages 95-96 in S. Bricker, B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner, editors. Effects of nutrient enrichment in the nation's estuaries : a decade of change. National Centers for Coastal Ocean Science, Silver Spring, MD.

Newton, J. and K. Van Voorhis. 2002. Seasonal patterns and controlling factors of primary production in Puget Sound's central basin and Possesion Sound. WA Department of Ecology, Olympia, WA.

Newton, J. A., S. L. Albertson, K. Van Voorhis, C. Maloy, and E. Siegel. 2002. Washington state marine water quality, 1998 through 2000. Washington State Department of Ecology, Olympia, WA.

Stark, K., S. Mickelson, and S. Keever. 2008. 2005 - 2007 Water Quality Status Report for Marine Waters. King County Department of Natural Resources and Parks, Seattle, WA.

U.S. Environmental Protection Agency. 2006. Puget Sound Georgia Basin ecosystem indicators report: technical background document. U.S. Environmental Protection Agency Region 10.

Washington State Department of Ecology. 2006a. Water Quality Program Policy: Assessment of water quality for the clean water act sections 303(d) and 305(b0 Integrated report. Washington State Department of Ecology, Olympia, WA.

Washington State Department of Ecology. 2006b. Water Quality Program Policy: Ensuring credible data for water quality management. Washington State Department of Ecology, Olympia, WA.

Marine Fecal Bacteria

Background

Fecal bacteria are found in the feces of humans and other homeothermic animals. They are monitored in recreational waters because they are good indicators of harmful pathogens that are more difficult to measure. The two types of fecal bacteria monitored in Puget Sound are fecal coliforms (including E. coli), which are gram-negative rod-shaped bacteria, and enterococci, which are gram-positive spherical bacteria. While fecal coliforms are more commonly monitored, enterococci are also measured because they have higher survival in salt water than coliforms and because they are thought to be more tightly associated with pathogens harmful to humans (Wymer et al. 2005). In Puget Sound, fecal pollution comes from both point-source origins such as combined sewer overflows and direct marine effluent discharge as well as non point-source origins such as surface water runoff, both of which increase with rainfall and river and stream discharge. In addition to serving as an indicator of pathogens, fecal bacterial pollution can also be an indicator of nutrient loading because sewage often contains high levels of nitrogen and phosphorous (Taslakian and Hardy 1976, Costanzo et al. 2001). Both point source (failing septic systems) and non-point sources (landscape features) contribute to fecal bacterial levels in Puget Sound. Additionally, shoreline and basin hydrology can affect the degree of retention of fecal coliform pollution such that bacteria may dissipate more slowly in enclosed bays with diminished water turnover. There currently are approximately 60 permitted wastewater treatment discharge locations in Puget Sound (Stark et al. 2009) (Figure 1) as well as numerous other storm drain and outfall locations.

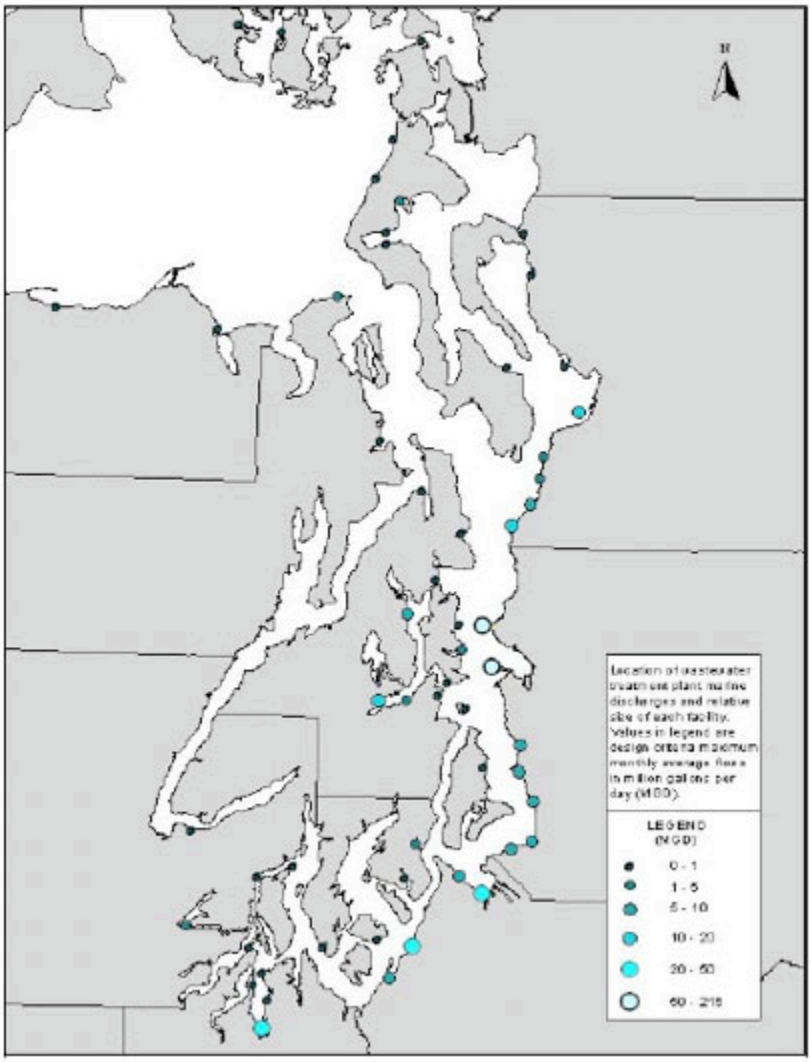


Figure 1. Puget Sound wastewater treatment plant marine discharge locations (reprinted form Stark et al. 2009 with permission from King County Department of Natural Resources and Parks).

In Puget Sound, monitoring of fecal bacteria is conducted by the Washington Department of Health, the Washington Department of Ecology and King County as part of the Puget Sound Ambient Monitoring Project (PSAMP) as well as other local municipalities. The Department of Ecology conducts monthly offshore surveys and assesses both fecal coliforms and enterococci at approximately 40 permanent stations along with a suite of locations that rotate each year (Janzen 1992, Newton et al. 2002)(Figure 2). The Department of Health (DOH) monitors fecal coliforms at 97 commercial shellfish growing areas in Puget Sound (Figure 3). The King County Department of Natural Resources and Parks monitors a combination of inshore and offshore targeted point-source (waste-water discharge) and ambient stations throughout central Puget Sound. The EPA-funded and jointly run (Departments of Health and Ecology) Beach

Environmental Assessment, Communication and Health (BEACH) program monitors and reports on enterococci levels at marine swimming beaches throughout Puget Sound.

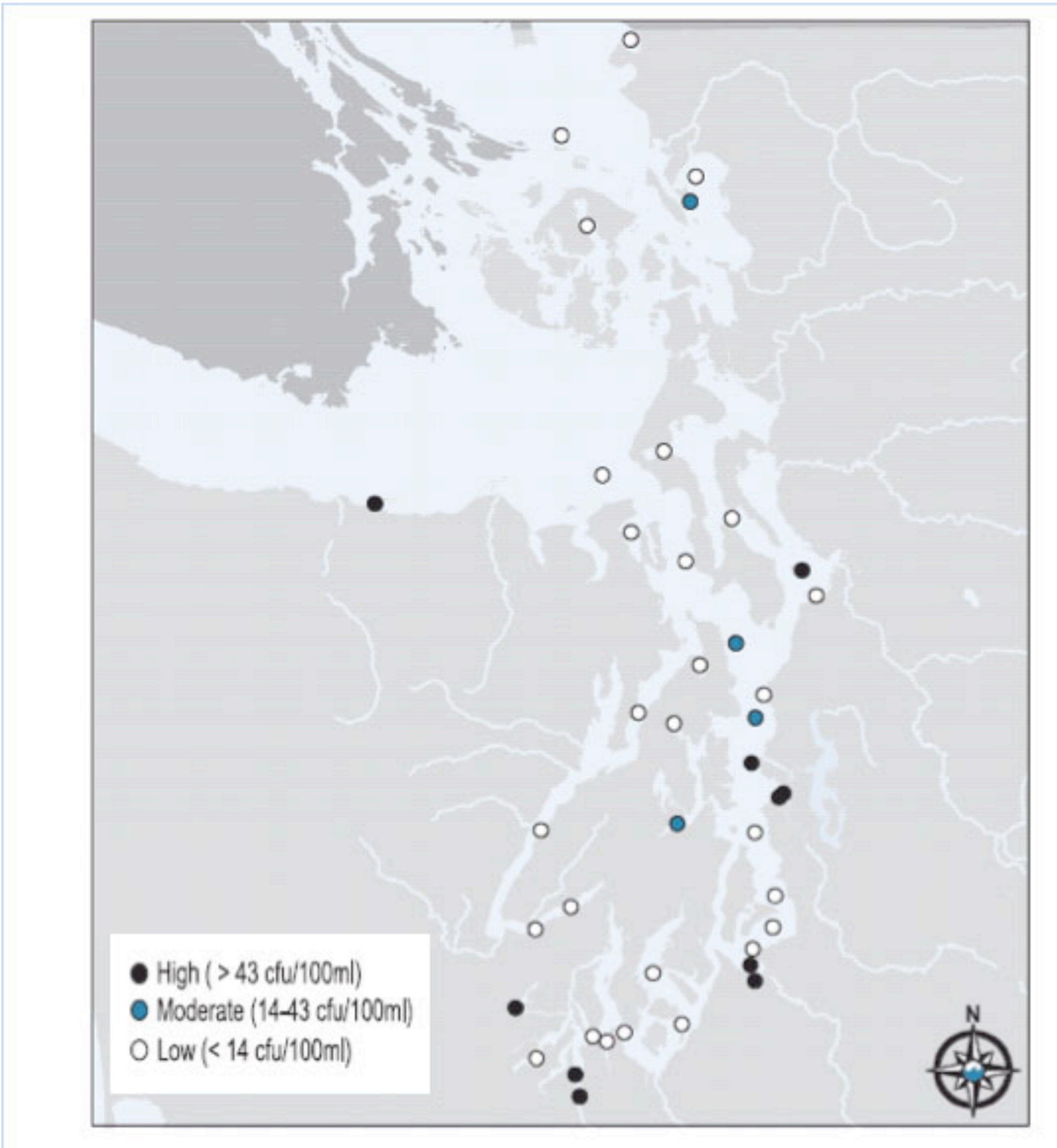


Figure 2. Department of Ecology Marine Waters monitoring stations and maximum fecal coliform bacteria levels (High, Moderate and Low detected Colony Forming Units) from 2001 – 2005 (reprinted from PSP 2007; methodology from Janzen 1992, Newton et al. 2002).

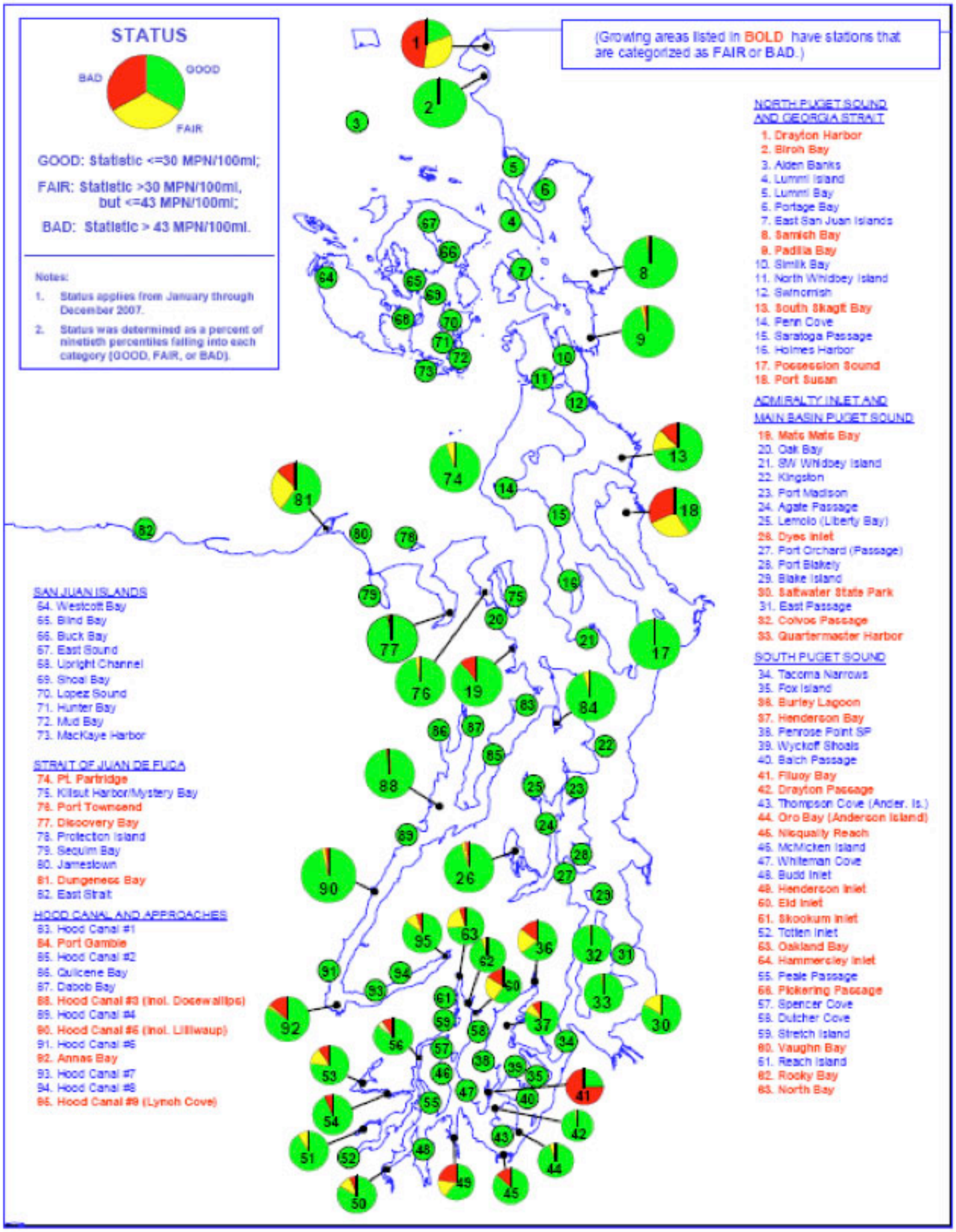

STATUS
BAD
GOOD
FAIR
GOOD: Statistic <=30 MPN/100ml;
FAIR: Statistic >30 MPN/100ml, but <=43 MPN/100ml;
BAD: Statistic > 43 MPN/100ml.
Notes:
1. Status applies from January through December 2007.
2. Status was determined as a percent of ninetieth percentiles falling into each category (GOOD, FAIR, or BAD).
(Growing areas listed in BOLD have stations that are categorized as FAIR or BAD.)
NORTH PUGET SOUND AND GEORGIA STRAIT
1. Drayton Harbor
2. Birch Bay
3. Alden Banks
4. Lummi Island
5. Lummi Bay
6. Portage Bay
7. East San Juan Islands
8. Samish Bay
9. Padilla Bay
10. Similk Bay
11. North Whidbey Island
12. Swinomish
13. South Skagit Bay
14. Penn Cove
15. Saratoga Passage
16. Holmes Harbor
17. Possession Sound
18. Port Susan
ADMIRALTY INLET AND MAIN BASIN PUGET SOUND
19. Mats Mats Bay
20. Oak Bay
21. SW Whidbey Island
22. Kingston
23. Port Madison
24. Agate Passage
25. Lemolo (Liberty Bay)
26. Dyes Inlet
27. Port Orchard (Passage)
28. Port Blakely
29. Blake Island
30. Saltwater State Park
31. East Passage
32. Colvos Passage
33. Quartermaster Harbor
SOUTH PUGET SOUND
34. Tacoma Narrows
35. Fox Island
36. Burley Lagoon
37. Henderson Bay
38. Penrose Point SP
39. Wyckoff Shoals
40. Balch Passage
41. Filucy Bay
42. Drayton Passage
43. Thompson Cove (Ander. Is.)
44. Oro Bay (Anderson Island)
45. Nisqually Reach
46. McMicken Island
47. Whiteman Cove
48. Budd Inlet
49. Henderson Inlet
50. Eld Inlet
51. Skookum Inlet
52. Totten Inlet
53. Oakland Bay
54. Hammersley Inlet
55. Peale Passage
56. Pickering Passage
57. Spencer Cove
58. Dutcher Cove
59. Stretch Island
60. Vaughn Bay
61. Reach Island
62. Rocky Bay
63. North Bay
SAN JUAN ISLANDS
64. Westcott Bay
65. Blind Bay
66. Buck Bay
67. East Sound
68. Upright Channel
69. Shoal Bay
70. Lopez Sound
71. Hunter Bay
72. Mud Bay
73. MacKaye Harbor
STRAIT OF JUAN DE FUCA
74. Pt. Partridge
75. Kilsut Harbor/Mystery Bay
76. Port Townsend
77. Discovery Bay
78. Protection Island
79. Sequim Bay
80. Jamestown
81. Dungeness Bay
82. East Strait
HOOD CANAL AND APPROACHES
83. Hood Canal #1
84. Port Gamble
85. Hood Canal #2
86. Quilcene Bay
87. Dabob Bay
88. Hood Canal #3 (incl. Dosewallips)
89. Hood Canal #4
90. Hood Canal #5 (incl. Lilliwaup)
91. Hood Canal #6
92. Annas Bay
93. Hood Canal #7
94. Hood Canal #8
95. Hood Canal #9 (Lynch Cove)

Figure 3. Commercial shellfish growing areas monitored by the Department of Health in 2007 with fecal pollution levels measured in Most Probable Number (MPN)/100m. Pie charts show the proportion of samples at each location with Good (≤ 30 MPN/100mL), Fair (>3 and ≤ 43 MPN/100mL) and Bad(> 43 MPN/100mL) fecal pollution levels (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

Monitoring by all agencies is conducted with the intent of determining whether bacterial counts meet or exceed established critical levels. For fecal coliforms, the State of Washington (WAC 173-201, 1991) mandates that in class A and AA marine waters, bacterial counts should not exceed a geometric mean of 14 organisms/100mL with no more than 10 % exceeding 43 organisms/100mL (Newton et al. 2002). Similar standards for coliforms are mandated by the National Shellfish Sanitation Program (NSSP) for shellfish growing areas such that the geometric mean of an area cannot exceed 14 organisms/100mL or that the estimated 90th percentile cannot exceed 43 organisms for cases where only non-point sources are present. For enterococci, the minimum advisory standard recommended by the EPA for recreational beaches is 35 colonies/100mL (Schneider 2002, Wymer et al. 2005). Fecal coliform levels are also a component of Federal Clean Water Act standards. Two agencies, the Department of Health (Determan 2009) and King County (Stark et al. 2009), have developed indices to rank sites according to the frequency and intensity of increases above Washington State standards in observed fecal coliform levels.

Status and Trends

The most recently reported assessment of fecal coliforms by the Department of Ecology monitoring program revealed that the highest levels of coliforms were observed in Budd Inlet, Commencement Bay, Oakland Bay, Port Angeles Harbor, Possession Sound and Elliot Bay from 2001 – 2005 (Janzen 1992, methodology from Newton et al. 2002, reported in PSP 2007)(Figure 2). Of the 97 shellfish growing areas tested by the Department of Health in 2007, the highest fecal pollution levels were found in Filucy Bay, Drayton Harbor, Burley Lagoon and Port Susan (Determan 2009)(Figures 3, 4). Using a calculated Fecal Pollution Index, which integrates the frequency and intensity of events of elevated fecal coliform levels and ranges from 1 to 3, they found that the sound-wide FPI was 1.16 (Determan 2009). A trend analysis showed that the sound-wide FPI had not changed significantly from 1998 – 2007 (Determan 2009)(Figure 5). The Frequency of Exceedence (FOE) index of fecal coliform bacteria utilized by the King County shellfish area monitoring program identified Alki Point, Shilshole Bay, Fauntleroy Cove, Magnolia and Inner Elliott Bay as the locations with the highest FOE in 2004 (reported in PSP 2007, methodology from Stark et al. 2009)(Figure 1). The most recent enterococci levels reported by the BEACH program showed that of the 70 beaches monitored in 2004 and 2005, the highest number of exceedances were in locations that were largely on septic systems such as Birch Bay County Park and Bayview State Park, in enclosed bays such as Freeland Park as well as beaches in Sinclair and Dyes Inlets and Twanoh State Park (methodology from Schneider 2004, reported in PSP 2007)(Figure 7).

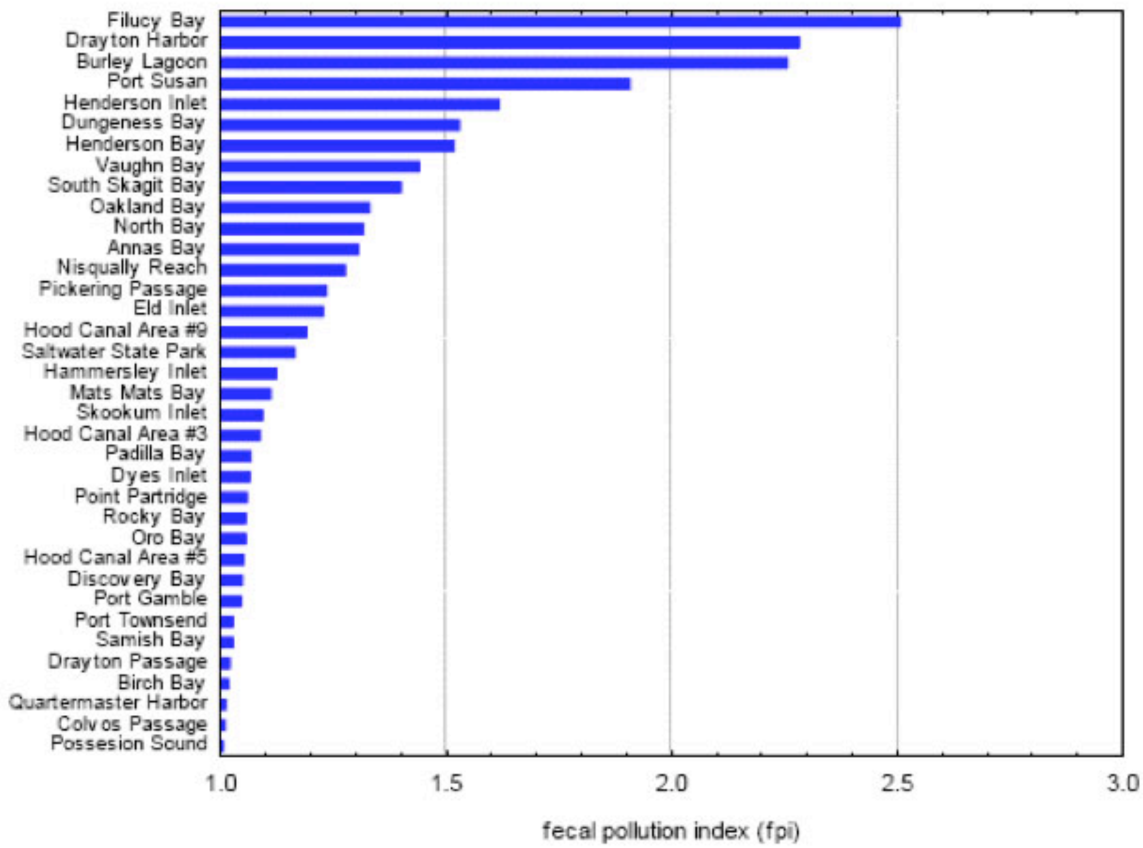


Figure 4. Department of Health rankings of 36 commercial shellfish growing areas in Puget Sound according to the fecal pollution index in 2007 (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

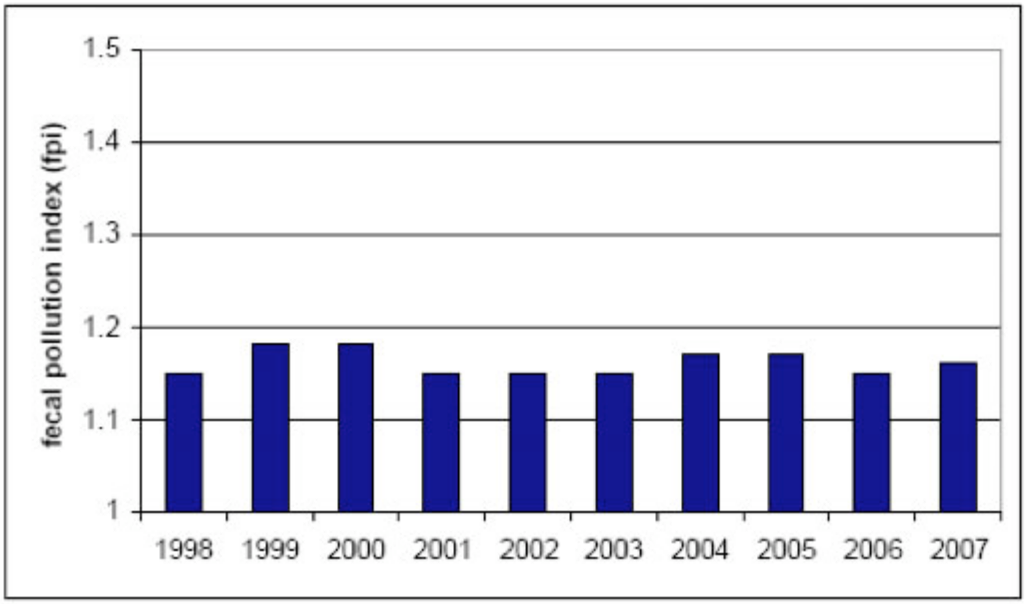


Figure 5. Fecal pollution index at commercial growing areas monitored by the Department of Health in Puget Sound from 1998 – 2007 (reprinted from Determan 2009; courtesy of Washington State Department of Health Shellfish Program).

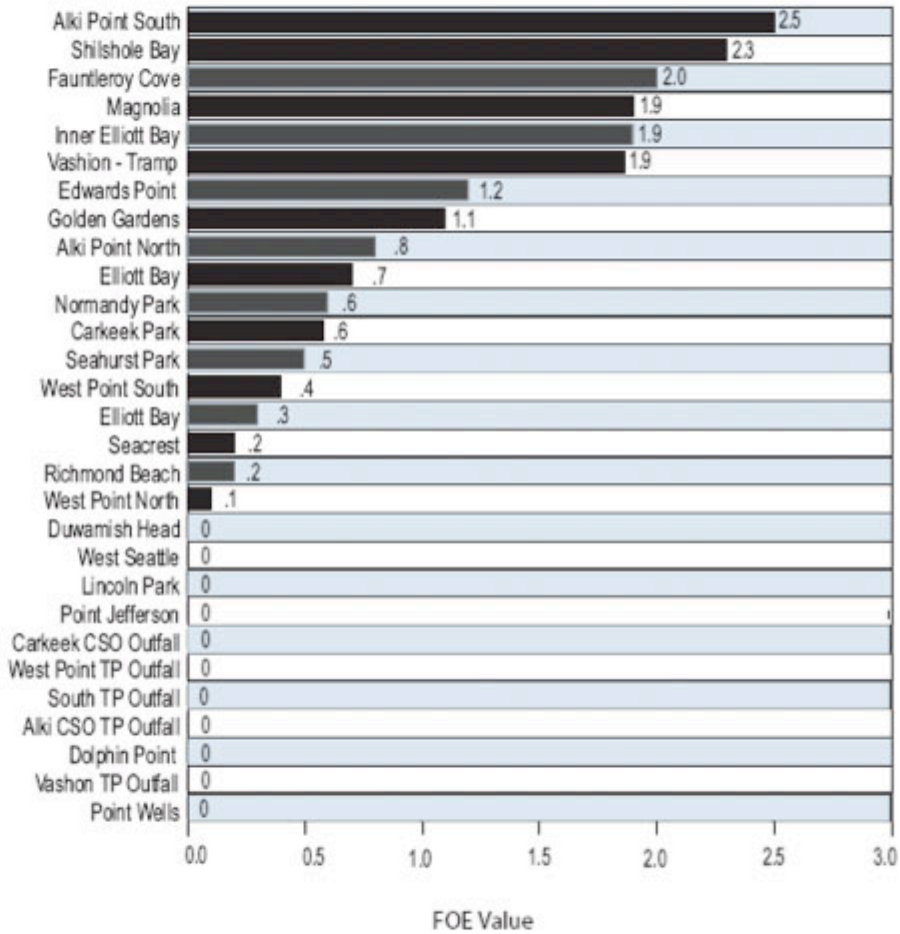


Figure 6. Frequency Of Exceedence (FOE) index of fecal coliform bacteria from offshore and beach stations monitored by King County Department of Natural Resources and Parks in 2004 (reprinted from PSP 2007; methodology from Stark et al. 2009).

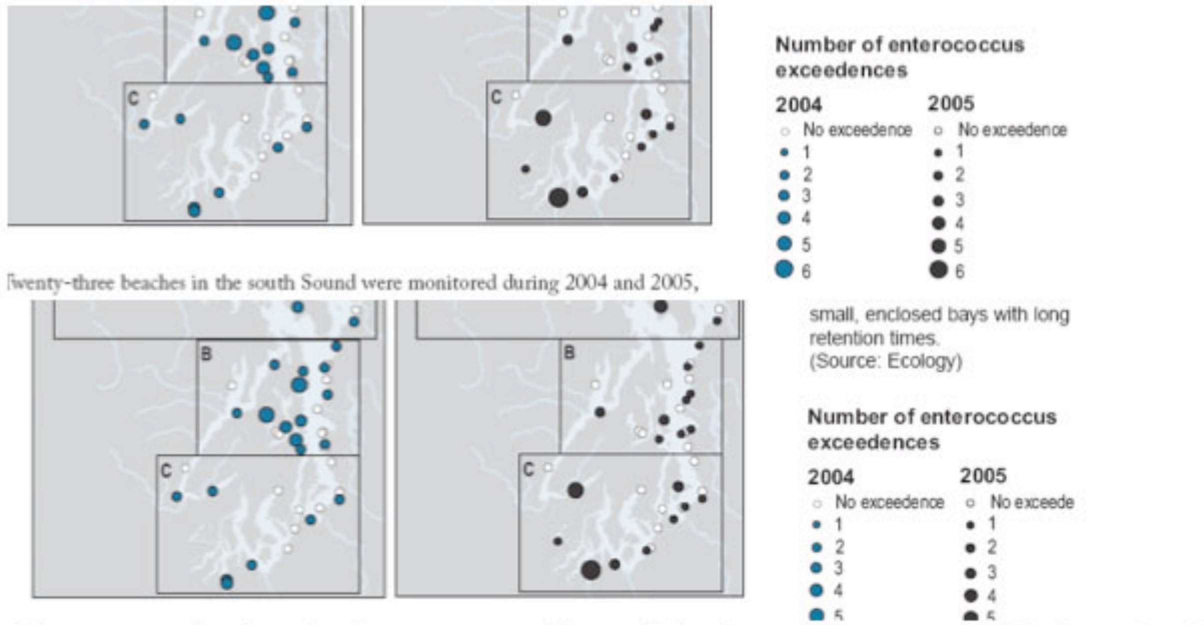


Figure 7. Monitoring sites for enterococci bacteria by the BEACH program (jointly run by the Department of Ecology and the Department of Health) and the number of times enterococci levels location exceeded program-defined guidelines (reprinted from PSP 2007; methodology from Schneider 2002, 2004).

Uncertainties

While fecal coliform levels in Puget Sound are well documented, disparate data sources make understanding broad spatial and temporal trends challenging, thereby obscuring potentially important patterns. Local hydrology, water temperature and salinity may all affect the persistence of fecal coliforms in Puget Sound yet this has not been investigated. Finally, the relative contribution of pet waste to overall fecal coliforms levels in Puget Sound has not been examined yet disease transfer from domestic pets to mammalian wildlife by this mechanism has been shown in other systems (Miller et al. 2002).

Summary

Considerable monitoring effort contributes to the assessment of fecal bacteria in Puget Sound. No single area or basin of Puget Sound was identified as consistently having the highest fecal coliform levels. A single analysis evaluating spatial and temporal trends based on all available data sources for fecal bacteria in Puget Sound has not been conducted.

Literature Cited

Costanzo, S. D., M. J. O'Donohue, W. C. Dennison, N. R. Loneragan, and M. Thomas. 2001. A new approach for detecting and mapping sewage impacts. Marine Pollution Bulletin 42:149-156.

Determan, T. 2009. Atlas of Fecal Coliform Pollution in Commercial Shellfish Areas of Puget Sound: Year 2007. Washington State Department of Health, Olympia, WA.

Janzen, C. 1992. Marine Water Column Ambient Monitoring Plan. Washington State Department of Ecology, Olympia, WA.

Miller, M. A., I. A. Gardner, C. Kreuder, D. M. Paradies, K. R. Worcester, D. A. Jessup, E. Dodd, M. D. Harris, J. A. Ames, A. E. Packham, and P. A. Conrad. 2002. Coastal freshwater runoff is a risk factor for *Toxoplasma gondii* infection of southern sea otters (*Enhydra lutris nereis*). International Journal for Parasitology 32:997-1006.

Newton, J. A., S. L. Albertson, k. V. Voorhis, C. Maloy, and E.Siegel. 2002. Washington State Marine Water Quality, 1998 through 2000. Washington State Department of Ecology, Olympia, WA.

PSP. 2007. Puget Sound Update: Ninth Report of the Puget Sound Assessment and Monitoring Program. Page 260, Seattle, WA.

Schneider, L. 2002. Draft BEACH Program Guidance. Washington State Department of Ecology, Olympia, WA.

Schneider, L. 2004. Quality Assurance Project Plan: BEACH Program. Washington State Department of Ecology, Olympia, WA.

Stark, K., S. Mickelson, and S. Keever. 2009. Water Quality Status Report for Marine Waters, 2005 - 2007. King County Department of Natural Resources and Parks, Seattle, WA.

Taslakian, M. J., and J. T. Hardy. 1976. Sewage nutrient enrichment and phytoplankton ecology along the central coast of Lebanon. Marine Biology 38:315-325.

Wymer, L. J., K. P. Brenner, J. W. Martinson, W. R. Stutts, S. A. Schaub, and A. P. Dufour. 2005. The EMPACT Beaches Project: Results from a study on microbiological monitoring in recreational waters. U.S. Environmental Protection Agency, Cincinnati, OH.

Water Quantity

Here we provide a limited synthesis of stream gauge data to examine trends in freshwater flows with respect to annual and daily flows, timing of flow, low flows and flows relative to instream flow guidelines. This is intended to supplement a review of published information, but we caution that a full analysis of these data and appropriate vetting of methods and interpretations is needed to fully assess the status of freshwater flows. It is our intention that this data compilation and analysis be used to identify data limitations and other key uncertainties with respect to the Puget Sound Partnership Water Quantity Priorities.

Data sources

There are approximately 90 gauging stations overseen by the United States Geological Survey (USGS) in the Puget Sound basin that are located on unregulated reaches of rivers and streams that may be suitable for the analysis of streamflow status and trends (United States Geological Survey 2010b). A complete analysis of all available data was not performed for this report. Instead, data from at least one unregulated gauging station within each Water Resource Inventory Area (WRIA) were included whenever possible. This selection was based on the intent to capture broad regional coverage.

We included all data from available gauging stations on unregulated reaches in the Skagit River basin in order to determine whether there were basin-wide correlations in the hydrologic indicators. Previous reports have combined streamflow data from several rivers to evaluate regional trends (Puget Sound Partnership 2009). A strong correlation between stream and rivers within the same basin could suggest that this is a valid approach. We review evidence for correlation here using simple descriptive statistics, but emphasize that a more rigorous analysis is warranted.

1. Flow Timing

Background

Puget Sound river hydrology could be affected by climate change. Precipitation in the region occurs predominately in the winter months. The accumulation of snow in the mountains is a primary storage mechanism, particularly for the snowmelt-dominated and transitional river systems. It has been estimated that more than 70% of total stream discharge in the Western United States is from melting snowpack (1996). An estimated 27% of summer streamflow of the Nooksack River originates from high-elevation snowshed and glacier-derived meltwater (Bach 2002). Climate change assessments predict increased winter and spring temperatures, resulting in decreased snowpack storage in the mountains, increased winter runoff as more precipitation falls as rain, and lower summer flows (Hamlet and Lettenmaier 1999, Lettenmaier et al. 1999, Mote et al. 1999, Leung et al. 2004, Barnett et al. 2008). Climate change may force rivers with snowmelt-dominated and transitional hydrological flow patterns towards rainfall-dominated hydrology (Mote et al. 1999). These changes are measurable through flow timing metrics, including the timing of the center of mass of annual flow (CT).

Prediction of the regional impacts of climate change on river and stream hydrology can be confounded by typical variation in rainfall patterns, high geographic variability, and land use changes. At least two large-scale systems affect annual climate variations in the Pacific Northwest (Mote 2003). These are the El Niño/Southern Oscillation, with a period of 2 to 7 years, and the Pacific Decadal Oscillation (PDO), with an estimated period of 20 to 30 years. Warm and cool phases of the El Niño/Southern Oscillation and/or Pacific Decadal Oscillation can result in variations on the order of 1°C for temperature, and 20% for precipitation (Mote et al. 2003). Hamlet et al. (2005) utilized a Variable Infiltration Capacity model to discern long-term trends in spring snowpack and snowmelt timing, decadal temperature and precipitation variability. They found that the date on which 90% snowmelt occurred correlated strongly with winter temperatures in the Pacific Northwest, and that there was a long-term warming trend that was not associated with decadal oscillations. In a subsequent study, Hamlet et al. (2007) specifically investigated the relationship between temperature, precipitation, and runoff timing in the western United States and found that in warmer areas, including the Pacific Northwest, fractional streamflow tended to occur earlier in the year (Hamlet et al. 2007). Mote et al. (2008) concluded that the primary factor in decreasing snowpack in the Washington Cascades was rising temperatures, consistent with the global warming. The long-term snowpack trends were unrelated to the variability brought about by Pacific oscillations.

Stewart et al. (2004) investigated historical (1948-2000) and projected future streamflow timing in snowmelt dominated rivers and streams in the Western United States. They found significant trends towards earlier runoff in many rivers and streams in the Pacific Northwest. Utilizing a 'business-as-usual' emissions scenario with a Parallel Climate Model, they predicted a continuation of this trend, largely due to increased winter and spring temperatures, but not changes in precipitation. In a companion study they further analyzed the trends in streamflow timing with variations of the PDO (Stewart et al. 2005). While streamflow timing was partially controlled by the PDO, there remained a significant part of the variation in timing that was explained by a longer-term warming trend in spring temperatures. This suggests that earlier seasonal flows may be associated with warming.

In addition to accelerated spring snowmelt, the shift toward earlier runoff timing can be attributed to a larger fraction of winter precipitation occurring as rain instead of snow. Knowles et al.(2006) evaluated data from the western United States and found a decreasing fraction of winter precipitation falling as snow. The largest decreases occurred in warmer winter areas, such as the Pacific Northwest, where relatively small increases in temperature would result in the transition from snowfall to rainfall, resulting in less snowpack and earlier runoff timing (Knowles et al. 2006).

Using a multivariate analysis, Barnett et al. (2008) evaluated simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999. They found significant increasing trends in winter temperature, and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt) and that this was mostly like driven by anthropogenic forcing (Barnett et al. 2008).

Recently, the Climate Impacts Group at the University of Washington performed The Washington Climate Change Impact Assessment. The assessment included analyses of hydrology and water resource management in which they utilized results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin they found a dramatic decrease in spring snowpack, with almost no April 1 snowpack by 2080. During that period, river hydrographs progressively changed from transition or snow-rain dominated to rain dominated patterns. There was little predicted change in annual precipitation.

The observed and predicted changes in river flow regime described above can affect water resource management in the Pacific Northwest where systems have been designed based on historical flow patterns (Lettenmaier et al. 1999, Milly et al. 2008). Wiley and Palmer(2008) utilized a three-stage modeling approach to evaluate the potential impacts of climate change on the Seattle water supply system. They found a decreasing annual system yield (the amount of water that can be reliably supplied by a system) largely due to earlier runoff and decreasing water storage in the mountain snowpack. Vano et al. (2009) expanded this analysis to include the Everett and Tacoma water systems. They found that altered flow regimes likely will reduce the reliability of all three systems, particularly in the face of increasing demand, and could affect ancillary operations such as flood control, power generation, and the augmentation of environmental flows.

Several measures of flow timing exist. One measure of river flow timing is centroid timing (CT), calculated by Stewart et al. (2005) and Elsner et al. (2009):

where: qi=daily flow and ti=number of days past the beginning of the water year.

The centroid of flow measure is relatively insensitive to false interannual variations, is easy to calculate, and allows for easy comparisons of basins (Stewart et al. 2004).

There are approximately 90 gauging stations overseen by the United States Geological Survey (USGS) in the Puget Sound basin that are located on unregulated reaches of rivers and streams, which may be suitable for the analysis of streamflow status and trends (USGS Water Center); a list is provided in Chapter 1 of this report. A complete analysis of all of the available data was not performed for the purposes of this report. However, data from at least one unregulated gauging station within each Water Resource Inventory Area (WRIA) was included where possible in order to coarsely approximate a regional scale.

Data from all available gauging stations on unregulated reaches in the Skagit River basin were included in this analysis in order to evaluate whether there existed any basin-wide correlations in the hydrologic indicators. Previous reports have combined streamflow data from several rivers to evaluate regional trends (PSP 2009). A strong correlation between stream and rivers within the same basin would indicate that this is a valid approach.

Status

Centroid timing values were calculated using gauge data from 14 different locations on unregulated rivers within the Puget Sound basin, in order to evaluate the status and trends of streamflow timing within the region. The results are shown in Table 1. The Pearson's Correlation Coefficients for the annual CT are shown in Table 2.

Table 1. Calculated centroid of flow timing (CT) and trends in CT for unregulated rivers and streams in the Puget Sound

			Centroid of Annual Flow		
River	Data Years		Average Date	Annual Change (days/year)	p (change ≠ 0)
WRIA 1 – Nooksack					
Nooksack USGS 12213100	1966-2009		3/18	-0.2±0.14	0.13
WRIA 3/4 – Upper-Lower Skagit and Samish					
Lower Sauk USGS 12189500	1936-2009		4/4	-0.20±0.08	0.01
Upper Sauk USGS 12186000	1929-2009		4/2	-0.17±0.08	0.03
Thunder USGS 12175500	1931-2009		5/16	-0.07±0.06	0.23
Newhalem USGS 12178100	1962-2009		4/10	-0.40±0.17	0.02
Samish USGS 12201500	1945-1970 1996-2009		2/13	-0.01±0.08	0.85
WRIA 5 - Stillaguamish					
Stillaguamish USGS 12167000	1929-2009		2/26	-0.13±0.06	0.05
WRIA 7 – Snohomish					
Skykomish USGS 12134500	1929-2009		3/22	-0.17±0.08	0.04
WRIA 8 – Cedar/Sammamish					
Cedar USGS 12114500	1947-2009		3/16	-0.15±0.13	0.23
WRIA 10 – Puyallup/White					
Puyallup USGS 12092000	1957-2009		4/6	0.01±0.13	0.94
WRIA 11 - Nisqually					
Nisqually USGS 12082500	1942-2009		3/30	-0.11±0.08	0.22
WRIA 13 - Deschutes					
Lower Deschutes USGS 12080010	1946-1963 1990-2009		2/22	0.00±0.07	0.97
Upper Deschutes USGS 12079000	1950-2009		2/14	0.02±0.09	0.80
WRIA 16 – Skokomish/Dosewalips					
Duckabush USGS 12054000	1939-2009		3/14	-0.11±0.09	0.20
Notes:	1. Center of Flow is calculated by: $CT = \frac{\sum(q_i t_i)}{\sum q_i}$				

Table 2. Pearson's Correlation Coefficient for annual CT for rivers within WRIA 3/4.

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.98	0.85	0.97	0.94	0.59
Upper Sauk			0.85	0.96	0.95	0.52
Thunder				0.88	0.85	0.54
Cascade					0.88	0.59
Newhalem						0.65

Note: All Peasrosn's correlation coefficients are significantly different than 0.

There appears to be a relatively strong correlation for this particular metric in flows within the Skagit River basin (r>0.85). The correlation between the rivers in the Skagit River Basin and the Samish River is less robust (r<0.65).

Trends

Annual CT values were calculated for the water years with complete data sets for 14 gauge stations in the Puget Sound. The trend of CT versus time was determined using simple linear regression. The significance of the trends were determined by evaluating the probability that the slope of the trendline was significantly different than zero. Results are shown in Table 1. The rivers with significant trends (P<0.05; Lower Sauk, Upper Sauk, Newhalem, NF Stillaguamish, and Skykomish) all showed an annual decrease in flow timing indicating that peak flows occur earlier in the year (Table 1). There were no rivers with significant trends indicating later flows. Overall, the centroid of flow timing occurred from 1.5-4 days earlier per decade. Data from two of the three rainfall-dominated river systems (Samish and Deschutes) and the single snowmelt-dominated river (Thunder) indicated no significant change in streamflow timing (P>0.05; Table 1).

Uncertainties

The analysis presented above was derived from data in the public domain. The values and trends for CT were calculated from average daily discharge data from USGS station located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted. Average daily discharge data for each water year (October 1 – September 30) were used to calculate the CT. The existence of trends was determined by evaluating the probability of the slope of the CT versus year, as determined through simple linear regression; trends were those with slope significantly different than zero (P<0.05).

Due to interannual variation, the selection of the beginning and ending years of streamflow data may affect the significance of the trend reported in Table 1 . Konrad et al. (2002) used both parametric and nonparametric tests and found a high likelihood of Type I errors when using 10-year streamflow records to evaluate long-term trends. In this evaluation we used a minimum record length of 37 years; the shortest record that resulted in a significant trend was 47 years.

The significance of the Pearson's correlation coefficient was determined by calculating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Of the fourteen data sets analyzed, four showed significant decreasing trends, indicating flow timing earlier in the water year. The rate of timing change was from 1.5-4 days per decade. The other ten data sets showed no significant trends.

There was significant variation in the flow timing data sets. However, there was a strong correlation in CT between rivers within the Skagit River basin (Pearson's r>0.85). The correlation between the CT of the Samish river and the rivers in the Skagit River basin was weaker (Pearson's r<0.65).

The CT could be a useful indicator of hydrologic alteration. It allows the tracking of potential changes due to climate, allows comparison of trends across different river systems, and is of importance to water resources managers. It may be more valuable when combined with other indicators of hydrologic alteration to give a more complete picture of streamflow patterns.

Literature Cited

Bach, A. 2002. Snowshed contributions to the Nooksack river watershed, North Cascades Range, Washington. Geographical Review 92:192-212.

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Cayan, D. R. 1996. Interannual climate variability and snowpack in the western United States. Journal of Climate 9:928-948.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F., and D. P. Lettenmaier. 1999. Effects of climate change on hydrology and water resources in the Columbia River basin. Journal of the American Water Resources Association 35:1597-1623.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate 20:1468-1486.

Knowles, N., M. D. Dettinger, and D. R. Cayan. 2006. Trends in snowfall versus rainfall in the Western United States. Journal of Climate 19:4545-4559.

Konrad, C. P., and D. Booth. 2002. Hydrologic Trends Associated with Urban Development for Selected Streams in the Puget Sound Basin, Western Washington. U.S. Geological Survey, Tacoma, WA.

Lettenmaier, D. P., A. W. Wood, R. N. Palmer, E. F. Wood, and E. Z. Stakhiv. 1999. Water resources implications of global warming: A US regional perspective. Climatic Change 43:537-579.

Leung, L. R., Y. Qian, X. D. Bian, W. M. Washington, J. G. Han, and J. O. Roads. 2004. Mid-century ensemble regional climate change scenarios for the western United States. Climatic Change 62:75-113.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science 319:573-574.

Mote, P., D. Canning, D. Fluharty, R. Francis, J. Franklin, A. Hamlet, M. Hershman, M. Holmberg, K. G. Ideker, W. Keeton, D. Lettenmaier, R. Leung, N. Mantua, E. Miles, B. Noble, H. Parandvash, D. W. Peterson, A. Snover, and S. Willard. 1999. Impacts of climate variability and change inthe Pacific Northwest. The JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Mote, P., A. Hamlet, and E. Salathe. 2008. Has spring snowpack declined in the Washington Cascades? Hydrology and Earth System Sciences 12:193-206.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

PSP. 2009. Ecosystem Status and Trends. Water Quantity., Puget Sound Partnership.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change 62:217-232.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington. http://waterdata.usgs.gov/wa/nwis/help/?provisional

United States Geological Survey. 2010b. USGS Washington Water Science Center. http://wa.water.usgs.gov/realtime/current.html

Vano, J. A., N. Voisin, L. Cuo, A. F. Hamlet, M. M. Elsner, R. N. Palmer, A. Polebitski, and D. P. Lettenmaier. 2009. Climate Change Impacts on Water Management in the Puget Sound

Region, Washington, USA. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Wiley, M. W., and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Average Annual Flow

Background

Average annual flow rate can be affected by changes in precipitation. Analysis of historical precipitation data suggests that significant trends in historical rainfall patterns associated with climate change in the Pacific Northwest are not detectable (Hamlet et al. 2005, Mote et al. 2005, Hamlet and Lettenmaier 2007, Hamlet et al. 2007). Climate change modeling suggests that there may be only modest increases in annual precipitation by 2080 (Elsner et al. 2009). Annual rainfall has been shown to be correlated with the Pacific Decadal Oscillation and El Nino Southern Oscillation, and variations in rainfall patterns may have increased in recent years (Hamlet and Lettenmaier 2007, Luce and Holden 2009). Increases in the variability of rainfall and streamflow in the Pacific Northwest may put pressure on water supply systems, which were designed based on historical variations (Jain et al. 2005, Hamlet and Lettenmaier 2007). One analysis (Pagano and Garen 2005) suggested that low-flow years were more likely to occur in succession, potentially exacerbating water supply pressures.

Luce and Holden (2009) utilized quartile regression to investigate trends in streamflow in wet (75th percentile), dry (25th percentile), and average (50th percentile) water years in rivers in the Pacific Northwest. They concluded that the dry years were getting dryer in the Pacific Northwest, accounting for much of the increased variability in annual streamflow.

Average annual flow may also be affected by land use changes. Logging in watersheds can reduce evapo-transpiration resulting in increased annual flows (Bosch and Hewlett 1982). Results from modeling studies suggest there is an increase in annual mean streamflow due to land use change in the Puget Sound lowlands (Cuo et al. 2009). The construction of storm drains associated with urbanization may result in lower streamflows (Simmons and Reynolds 1982). Increased diversions and consumptive uses may also result in lower overall streamflows.

Status and Trends

Data from the Cedar River (below Bear Creek, near Cedar Falls) indicated a significant decrease in annual average streamflow from 1946-2009 ($p=0.03$; ca. 0.3% yr-1 decrease; Table 1). No other river systems showed a significant change in annual average streamflow (Table 1). The Pearson's Correlation Coefficients for the average annual flow rate between the river systems in WRIA 3/4 indicate that there is a strong linear correlation between the annual average flow rates of the rivers evaluated ($r>0.83$; Table 2). There was a somewhat weaker correlation ($0.68<r<0.81$) between the Samish River and the rivers of the Skagit River basin,,all of which lie within WRIA 3/4.

Table 1. Average annual flow rate in cubic feet per second (CFS) and annual change in average flow rate as determined by simple linear regression (±standard error). Data from USGS Washington Water Science Center (http://wa.water.usgs.gov/)

			AVERAGE FLOW	
River	**Data Years**		**Average Flow Rate**	**Annual Change**
			(CFS)	(ΔCFS/Year)
WRIA 1 – Nooksack				
Nooksack USGS 12213100	1966-2009		3855	-3.7±9.0
WRIA 3/4 – Upper-Lower Skagit and Samish				
Lower Sauk USGS 12189500	1936-2009		4342	2.0±4.5
Upper Sauk USGS 12186000	1929-2009		1118	0.0±1.1
Thunder USGS 12175500	1931-2009		619	0.2±0.4
Newhalem USGS 12178100	1962-2009		176	0.1±0.3
Samish USGS 12201500	1945-1970 1996-2009		246	0.2±0.4
WRIA 5 - Stillaguamish				
Stillaguamish USGS 12167000	1929-2009		1897	2.8±1.9
WRIA 7 – Snohomish				
Skykomish USGS 12134500	1929-2009		3957	3.5±4.1
WRIA 8 – Cedar/Sammamish				
Cedar USGS 12114500	1947-2009		161	-0.5±0.2
WRIA 10 – Puyallup/White				
Puyallup USGS 12092000	1957-2009		527	-0.4±0.6
WRIA 11 - Nisqually				
Nisqually USGS 12082500	1942-2009		772	-0.0±0.9
WRIA 13 - Deschutes				
Lower Deschutes USGS 12080010	1946-1963 1990-2009		397	0.2±0.9
Upper Deschutes USGS 12079000	1950-2009		258	-0.2±0.7
WRIA 16 – Skokomish/Dosewalips				
Duckabush USGS 12054000	1939-2009		416	0.0±0.5

Table 2. Pearson's Correlation Coefficient of annual average flow rates between river systems in WRIA 3/4. All correlations are significantly different than zero ($P<0.05$).

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.98	0.85	0.97	0.94	0.81
Upper Sauk			0.83	0.97	0.94	0.75
Thunder				0.87	0.86	0.68
Cascade					0.87	0.73
Newhalem						0.73

Uncertainties

This analysis was derived from data within the public domain. Average annual flow data presented were calculated from average daily discharge data from USGS stations located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

Average daily discharge data for each water year (October 1 – September 30) were used to calculate annual average flow rates. Trends were determined by evaluating the probability that the slope of the average annual flow versus year, as determined through simple linear regression, was significantly different than zero ($p<0.05$).

The significance of the Pearson's correlation coefficient was determined by calculating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Of the 14 locations analyzed, only one showed a significant change in overall annual flow. All other results were not significant ($p>0.10$). Annual Average Flow rates are informative when used in combination with other hydrologic indicators such as summer low flows and indicator of flow timing.

Literature Cited

Bosch, J. M. and J. D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapo-transpiration. Journal of Hydrology 55:3-23.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the

Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F. and D. P. Lettenmaier. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research 43.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate 20:1468-1486.

Jain, S., M. Hoerling, and J. Eischeid. 2005. Decreasing reliability and increasing synchroneity of western North American streamflow. Journal of Climate 18:613-618.

Luce, C. H. and Z. A. Holden. 2009. Declining annual streamflow distributions in the Pacific Northwest United States, 1948-2006. Geophysical Research Letters 36.

Mote, P. W., A. F. Hamlet, M. P. Clark, and D. P. Lettenmaier. 2005. Declining mountain snowpack in western north America. Bulletin of the American Meteorological Society 86:39-+.

Pagano, T. and D. Garen. 2005. A recent increase in western US streamflow variability and persistence. Journal of Hydrometeorology 6:173-179.

Puget Sound Partnership. 2009. Ecosystem Status and Trends. Water Quantity., Puget Sound Partnership.

Simmons, D. L. and R. J. Reynolds. 1982. Effects of urbanizatsion on base-flow of selected south-shore streams, Long Island, New York. Water Resources Bulletin 18:797-805.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Average Daily Flow

Background

Streamflow patterns in Puget Sound rivers and streams are classified into three hydrologic regimes: snowmelt dominated, rainfall dominated, and transitional (Stewart et al. 2005, Beechie et al. 2006, Elsner et al. 2009). Generally, in snowmelt-dominated rivers, a significant proportion of winter precipitation is stored as snowpack, resulting in low winter flows with peak flows during the spring snowmelt period from April through July. Rainfall-dominated rivers experience peak flow during the winter months as the majority of precipitation falls as rain. Transitional rivers experience both winter and spring peak flows resulting from winter precipitation and spring snowmelt. Hydrologic flow regimes in Puget Sound rivers have been altered through the construction of dams for flood control or power generation, or by changes in land cover and climate. Alteration of historical flow patterns can cause ecological harm and disrupt supply (Poff et al. 1997, Wiley and Palmer 2008).

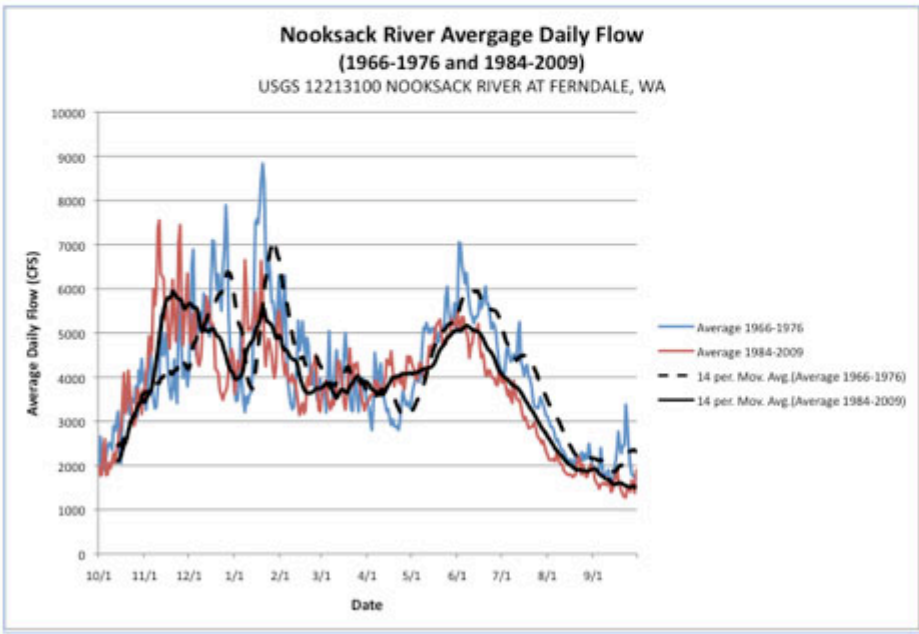
Barnett et al. (2008) utilized a multivariate analysis to evaluate simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999. They found significant increasing trends in winter temperature and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt). In order to distinguish natural variation from anthropogenic forcing, they evaluated the observations against two separate climate models and found that the hydrologic changes were both detectable and attributable to anthropogenic forcings.

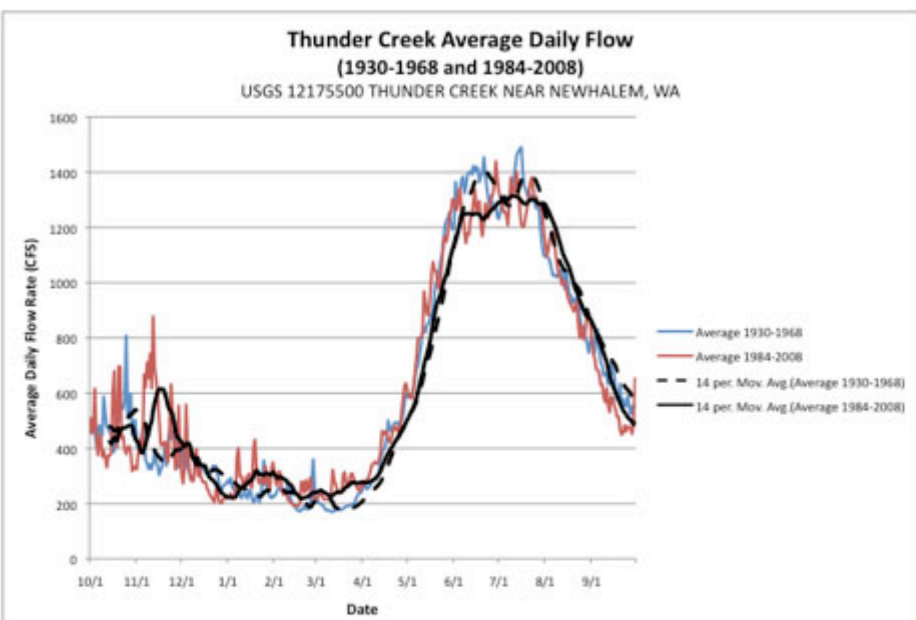
Stewart et al. (2004) investigated historic (1948-2000) and future streamflow timing in snowmelt dominated rivers and streams in the Western United States. They found significant trends towards earlier runoff in many rivers and streams in the Pacific Northwest. Utilizing a 'business-as-usual' emissions scenario with a Parallel Climate Model, they predicted continuation of this trend, due largely to increased winter and spring temperatures but not changes in precipitation. In a companion study they further analyzed the trends in streamflow timing with variations of the PDO (Stewart et al. 2005). While streamflow timing was partially controlled by the PDO there remained a substantial portion of the variation in timing that was explained by a longer-term warming trend in spring temperatures.

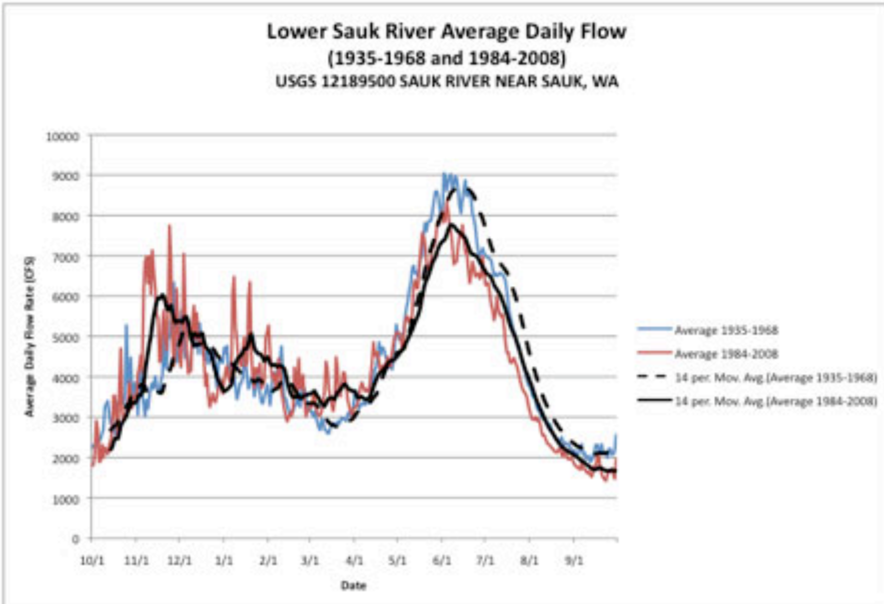
The Climate Impact Group at the University of Washington performed The Washington Climate Change Impact Assessment. The assessment included analyses of hydrology and water resource management utilizing results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin, they found a dramatic decrease in spring snowpack with there being almost no April 1 snowpack by 2080. Change in snowpack was correlated with a predicted change in river hydrography, from transition- or snow-rain dominated, to rain dominated patterns. There was little predicted change in annual precipitation.

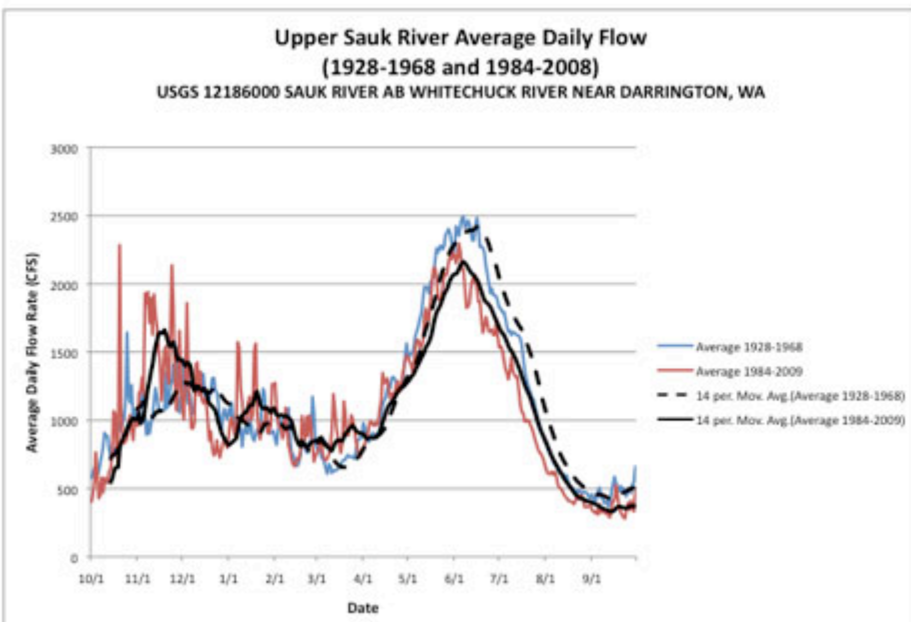
Status and Trends

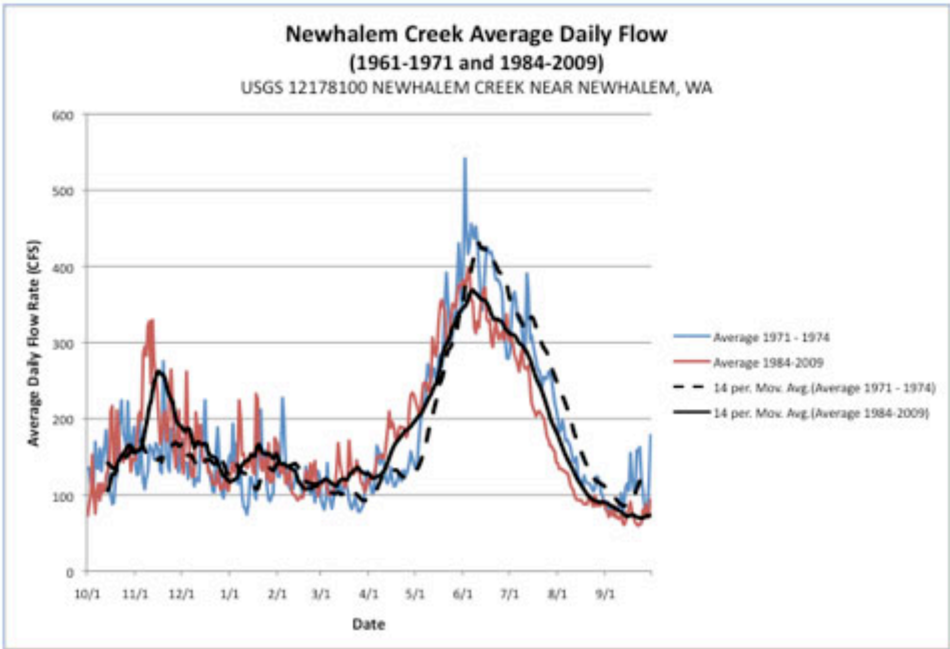
River hydrographs showing average annual daily flow from the initiative of observation through 1968, and from 1984 through 2009 are presented in Figure 1. Much of the warming trend observed in the Pacific Northwest has occurred since 1975 (Hamlet and Lettenmaier 2007). Comparing the streamflow patterns before and after this period could indicate effects of climate change.



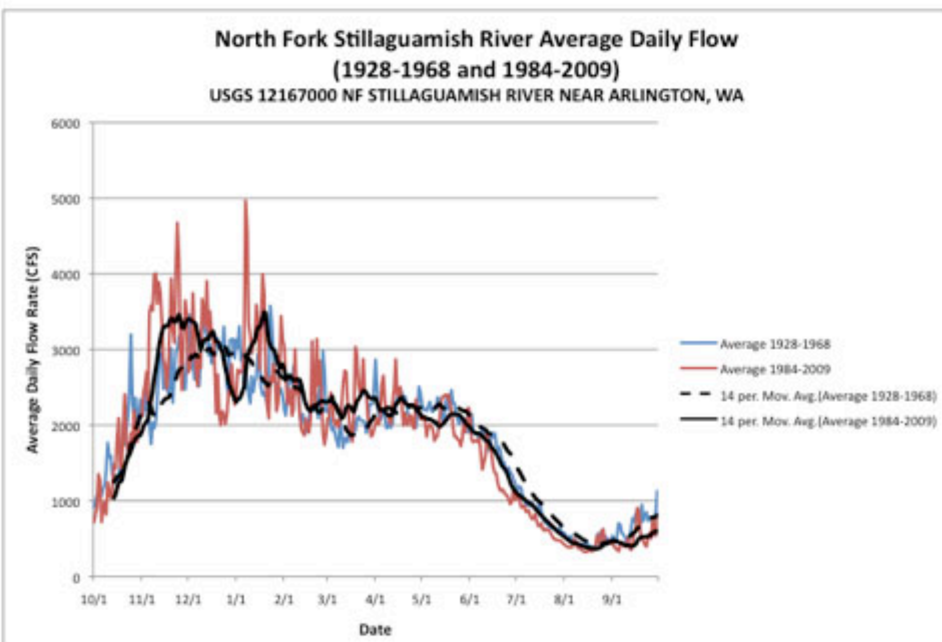



Lower Sauk River Average Daily Flow
(1935-1968 and 1984-2008)
USGS 12189500 SAUK RIVER NEAR SAUK, WA
Average Daily Flow Rate (CFS)
10000
9000
8000
7000
6000
5000
4000
3000
2000
1000
0
10/1
11/1
12/1
1/1
2/1
3/1
4/1
5/1
6/1
7/1
8/1
9/1
Date
Average 1935-1968
Average 1984-2008
14 per. Mov. Avg.(Average 1935-1968)
14 per. Mov. Avg.(Average 1984-2008)

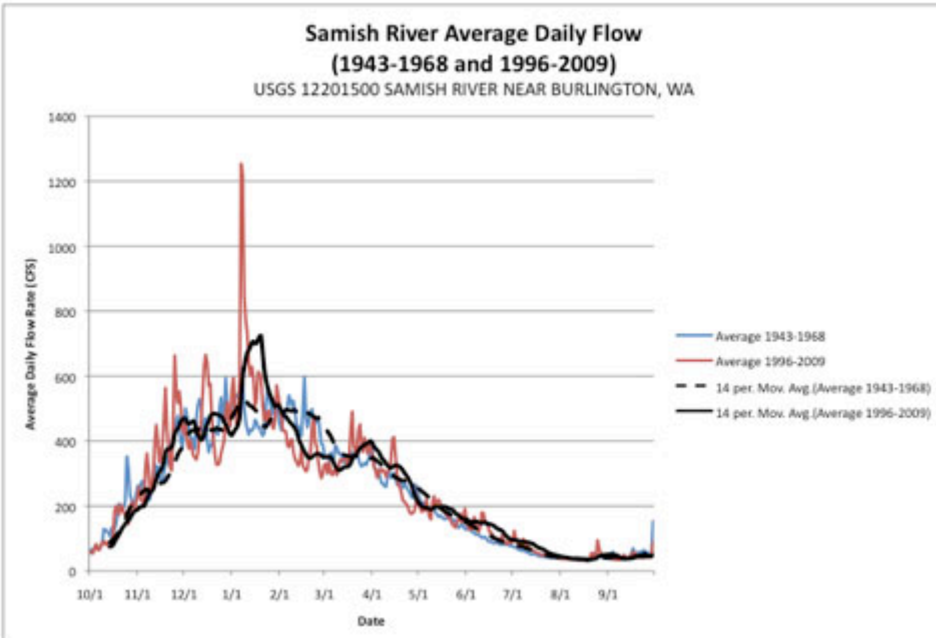

Upper Sauk River Average Daily Flow
(1928-1968 and 1984-2008)
USGS 12186000 SAUK RIVER AB WHITECHUCK RIVER NEAR DARRINGTON, WA
Average Daily Flow Rate (CFS)
3000
2500
2000
1500
1000
500
0
10/1
11/1
12/1
1/1
2/1
3/1
4/1
5/1
6/1
7/1
8/1
9/1
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)

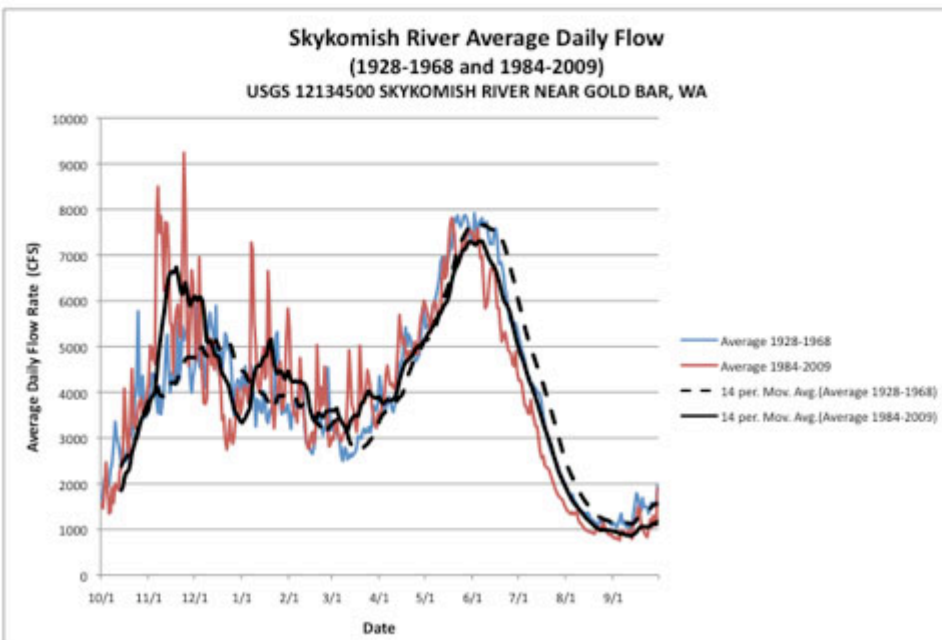


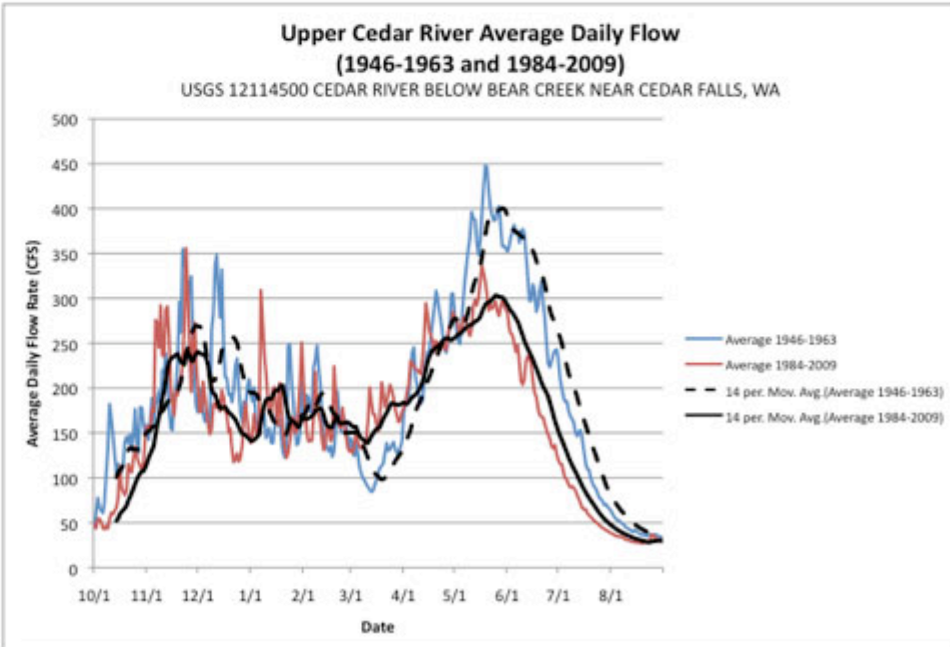
Newhalem Creek Average Daily Flow
(1961-1971 and 1984-2009)
USGS 12178100 NEWHALEM CREEK NEAR NEWHALEM, WA
Average Daily Flow Rate (CFS)
Date
Average 1971 - 1974
Average 1984-2009
14 per. Mov. Avg.(Average 1971 - 1974)
14 per. Mov. Avg.(Average 1984-2009)



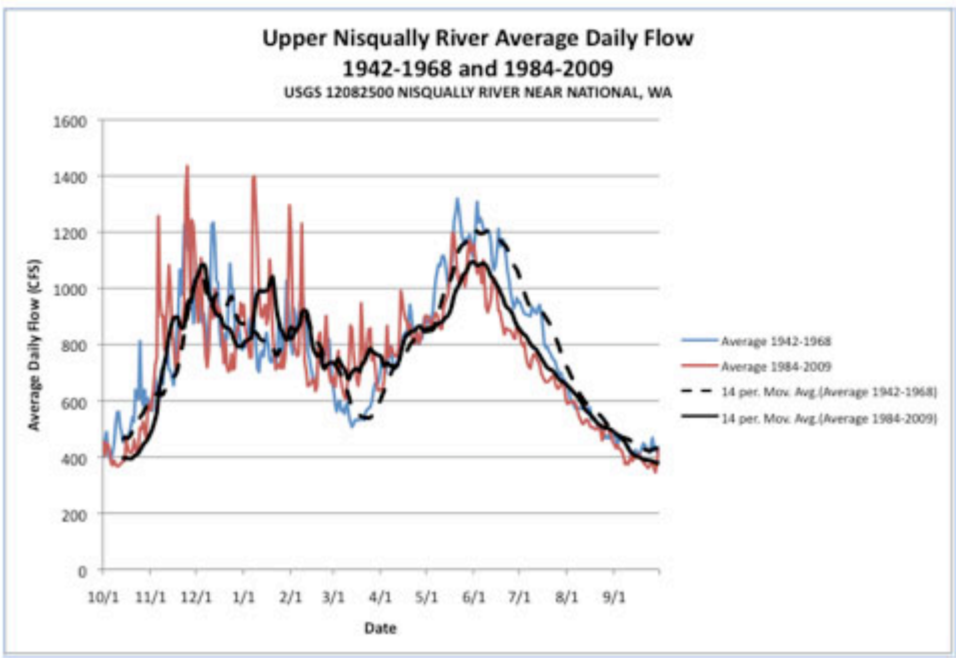
North Fork Stillaguamish River Average Daily Flow
(1928-1968 and 1984-2009)
USGS 12167000 NF STILLAGUAMISH RIVER NEAR ARLINGTON, WA
Average Daily Flow Rate (CFS)
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)


Samish River Average Daily Flow
(1943-1968 and 1996-2009)
USGS 12201500 SAMISH RIVER NEAR BURLINGTON, WA
Average Daily Flow Rate (CFS)
Date
Average 1943-1968
Average 1996-2009
14 per. Mov. Avg.(Average 1943-1968)
14 per. Mov. Avg.(Average 1996-2009)

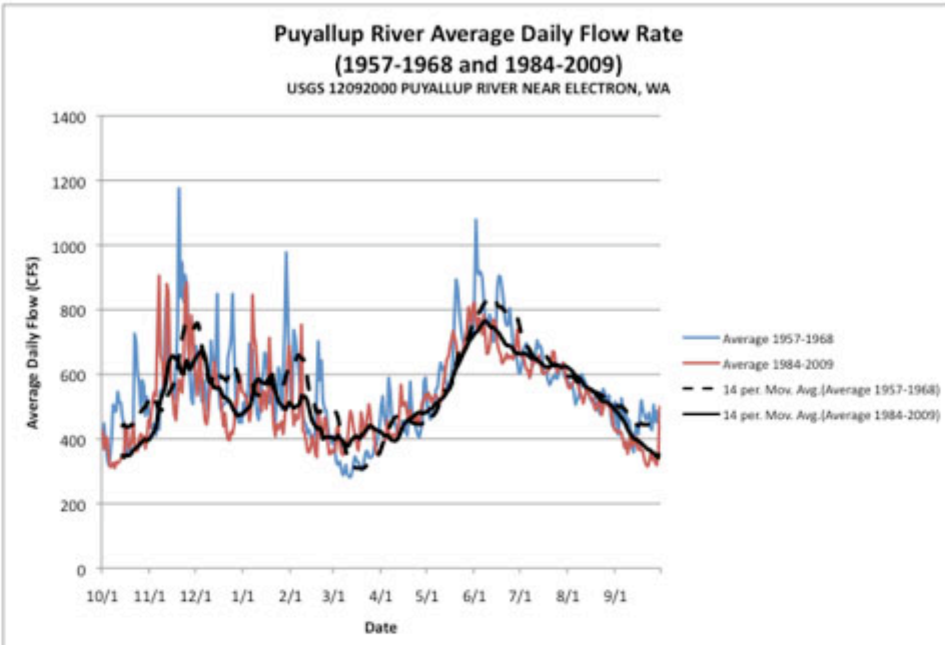

Skykomish River Average Daily Flow
(1928-1968 and 1984-2009)
USGS 12134500 SKYKOMISH RIVER NEAR GOLD BAR, WA
Average Daily Flow Rate (CFS)
Date
Average 1928-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1928-1968)
14 per. Mov. Avg.(Average 1984-2009)



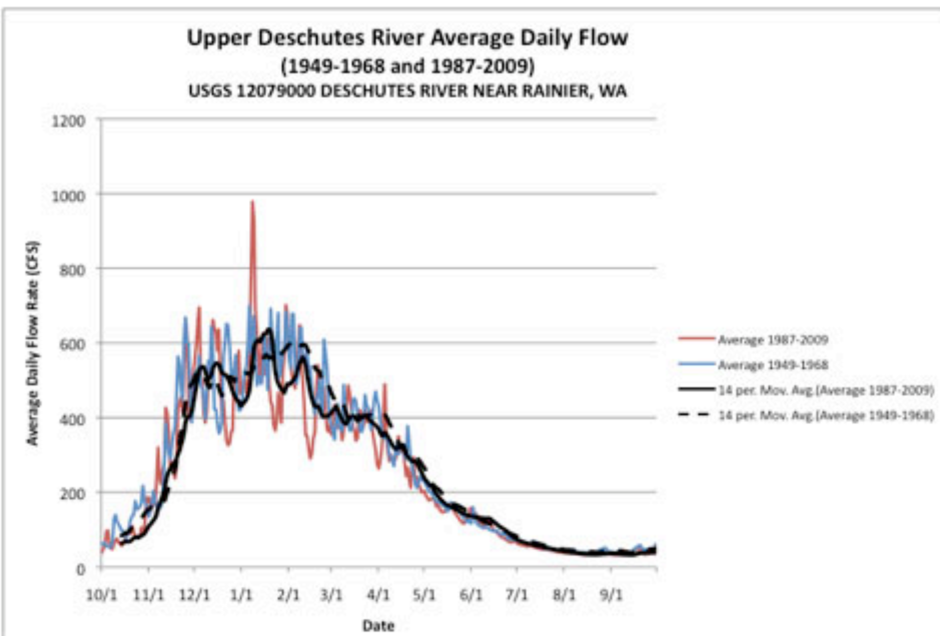
Upper Cedar River Average Daily Flow
(1946-1963 and 1984-2009)
USGS 12114500 CEDAR RIVER BELOW BEAR CREEK NEAR CEDAR FALLS, WA
Average Daily Flow Rate (CFS)
500
450
400
350
300
250
200
150
100
50
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1
Date
Average 1946-1963
Average 1984-2009
14 per. Mov. Avg.(Average 1946-1963)
14 per. Mov. Avg.(Average 1984-2009)



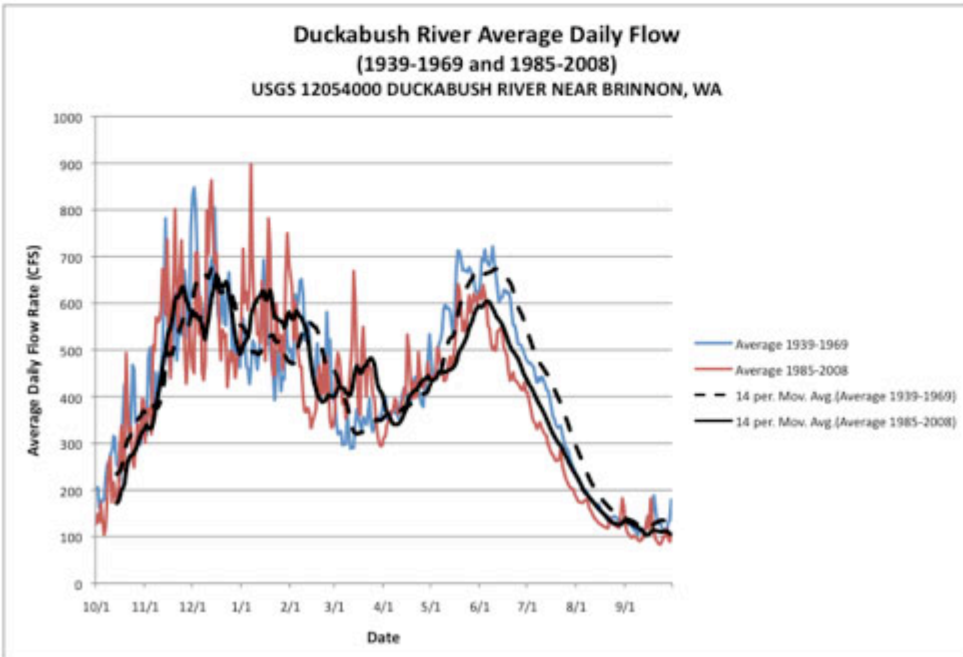
Upper Nisqually River Average Daily Flow
1942-1968 and 1984-2009
USGS 12082500 NISQUALLY RIVER NEAR NATIONAL, WA
Average Daily Flow (CFS)
1600
1400
1200
1000
800
600
400
200
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1942-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1942-1968)
14 per. Mov. Avg.(Average 1984-2009)



Puyallup River Average Daily Flow Rate
(1957-1968 and 1984-2009)
USGS 12092000 PUYALLUP RIVER NEAR ELECTRON, WA
Average Daily Flow (CFS)
1400
1200
1000
800
600
400
200
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1957-1968
Average 1984-2009
14 per. Mov. Avg.(Average 1957-1968)
14 per. Mov. Avg.(Average 1984-2009)



Upper Deschutes River Average Daily Flow
(1949-1968 and 1987-2009)
USGS 12079000 DESCHUTES RIVER NEAR RAINIER, WA
Average Daily Flow Rate (CFS)
1200
1000
800
600
400
200
0
10/1 11/1 12/1 1/1 2/1 3/1 4/1 5/1 6/1 7/1 8/1 9/1
Date
Average 1987-2009
Average 1949-1968
14 per. Mov. Avg.(Average 1987-2009)
14 per. Mov. Avg.(Average 1949-1968)



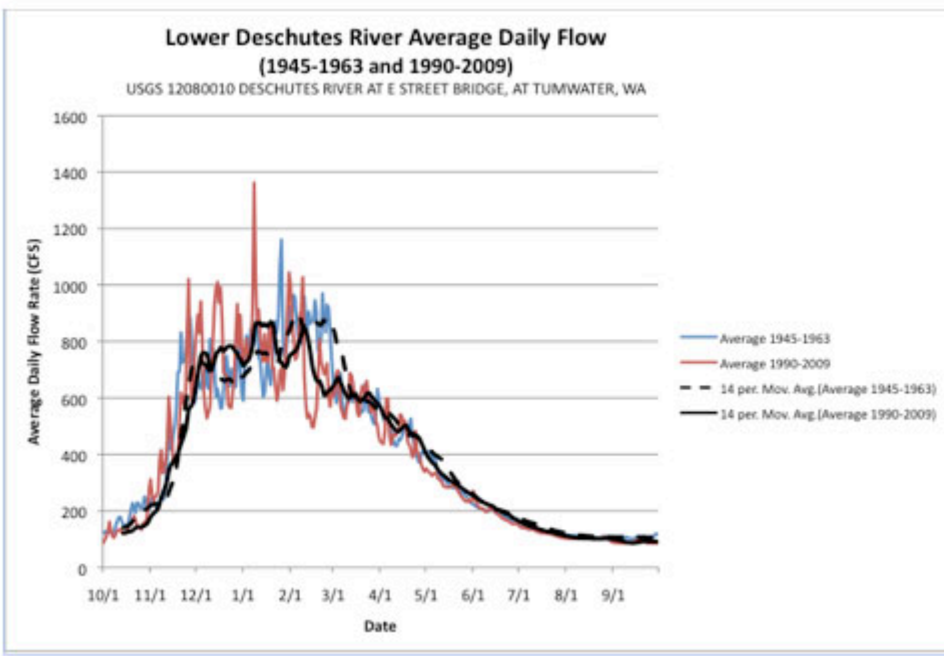


Figure 1. Average daily flow shown for historic (pre 1970's) and recent (mid-1980s to present) time period for 14 Puget Sound rivers. The time period varies slightly between river systems based on availability of data. Colored lines show average daily flow averaged over time period indicated in each of the chart title. Dark lines are 14-day smoothed averages for historic (dashed) or recent (solid) time periods. Data taken from United States Geological Service.

There is considerable variation even in averaged data which makes the detection of long-term trends problematic. However, the following generalities emerge. First, there has been little change in hydrologic patterns in rainfall-dominated rivers (Samish, Stillaguamish, and Deschutes) or in the snowmelt-dominated river (Thunder Creek). It is possible that consistent glacier melt contributed to the stable patterns in the latter river. However, there was an observable decline in spring peak flows in all of the transitional rivers (Nooksack, Sauk, Newhalem Creek, Skykomish, Upper Cedar, Upper Puyallup, Upper Nisqually, and Duckabush). Moreover, there appears to be a decline in the magnitude of the summer 7-day average low flows.

Uncertainties

The analysis presented above was derived from data in the public domain. Hydrographs were created utilizing average daily discharge data from USGS stations located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

The analysis in this section is qualitative and intended to illustrate potential changes in streamflow patters over time. Consequently, statistical significance was not determined. Specific streamflow measures, such as annual 7-day average low flow, or centroid of flow timing, are quantitative measures that can be evaluated statistically and are presented elsewhere in this document.

Summary

There is some evidence for changes in transitional river systems over time, indicated primarily as decreasing magnitude of the spring snowmelt peak flows. This is consistent with published predictions for the western North America. There also appears to be a decrease in the magnitude of summer low flows in transitional river systems. There was less evidence for change in daily flow patterns for rainfall-dominated or snowmelt-dominated river systems. Because of variation in hydrologic alteration, particularly between rivers or streams of differing classifications, combining streamflow information across multiple streams to evaluate general status and trends may not be appropriate and results should be interpreted with caution.

Literature Cited

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Hamlet, A. F. and D. P. Lettenmaier. 2007. Effects of 20th century warming and climate variability on flood risk in the western U.S. Water Resources Research 43.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47:769-784.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2004. Changes in snowmelt runoff timing in western North America under a 'business as usual' climate change scenario. Climatic Change 62:217-232.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Wiley, M. W. and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Seven-day Average Low Flow

Background

The hydrologic regime of rivers and streams in the Puget Sound is characterized by peak flows during the winter as a result of heavy precipitation, or during the spring due to snowmelt runoff. Base flows during the summer are low, consisting mainly of groundwater discharge. Base flows can be affected by climate change, urbanization, or groundwater withdrawals. Summer base flow levels are important ecologically because they can define or limit the availability of habitats. Summer base flow levels are important to water resource managers because low flows often coincide with peak consumption.

Climate change is expected to alter river hydrology in the Puget Sound basin. Observed and predicted increases in winter temperatures could result in more precipitation falling as rain instead of snow, earlier snowmelt timing, earlier streamflow timing, and lower summer flows (Mote et al. 1999, Mote et al. 2003). Several studies have evaluated the impacts of climate change on spring snowpack in the Pacific Northwest, with the conclusion that decreasing spring snowpack may result in lower summer flows. Long-term decline in snowpack in the Pacific Northwest was found to correlate largely with increasing temperatures, but not precipitation (Mote 2003). Follow-on studies with a Variable Infiltration Capacity model were performed to discern long term trends in spring snowpack from temperature and precipitation variability (Hamlet et al. 2005, Mote et al. 2005). Results suggested that long-term downward trends in spring snowpack were associated with widespread warming. Trends in snowpack associated with precipitation were largely controlled by decadal oscillations. Multiple regression analysis indicated that climatic oscillations accounted for approximately 10-60% of the trends in spring snowpack, depending on the time series examined (Mote 2006), leading the authors to conclude that the primary factor driving declining snowpack in the Washington Cascades was rising temperatures. The long-term snowpack trends were unrelated to the variability caused by Pacific oscillations.

Casola et al. (2009) investigated the potential impacts of climate change on snowpack by combining future temperature predictions with the estimated temperature sensitivity of spring snowpack. Analysis of historic and projected temperature data indicated that snowpack reductions over the past 30 years ranged from 8%-16% while future temperature change would result in an 11%-21% reduction in spring snowpack by 2050.

Stewart et al. (2005) evaluated the monthly fractional flow in snowmelt-dominated river systems in the Western United States and found an increasing fraction of flow occurring in March, corresponding with a decreasing fraction in June. Changes in streamflow pattern were associated with long-term increases in spring and winter temperatures, which spanned the decadal-scale Pacific climate oscillations. Barnett et al. (2008) utilized a multivariate analysis to evaluate the simultaneous changes in average winter temperature, snow pack, and runoff timing in the Western United States (including the Washington Cascades) for the period from 1950 – 1999 They found significant increasing trends in winter temperature, and decreasing trends in snow pack and runoff timing (indicating earlier snowmelt) associated with anthropogenic forcings.

The Climate Impacts Group utilized results from 20 global climate models and two emissions scenarios from the IPCC Special Report on Emissions Scenarios (A1B and B1) to evaluate projected changes in spring snowpack and runoff (Elsner et al. 2009). For the rivers in the Puget Sound basin they projected a dramatic decrease in spring snowpack with almost no April 1 snowpack by 2080. The climate change-related alterations in spring snowpack and streamflow timing are expected to result in lower summer flows.

Land use alterations can also result in lower summer flows. Urbanization and development are associated with an increase in impervious surface resulting in higher runoff fractions and lower infiltration (Burges et al. 1998). Reduced infiltration can lead to lower base flows, although this effect can be somewhat offset by a reduction in evapo-transpiration from the clearing of trees (Cuo et al. 2008). The construction of storm drain systems has been implicated as a primary factor in the reduction a base flows (Simmons and Reynolds 1982).

Cuo et al. (2009) utilized a Distributed Hydrology-Soil-Vegetation Model in order to determine the relative effects of land cover and temperature change on flow patterns in Puget Sound streams. They found that the relative importance of temperature and land cover differed between the upland and lowland basins. In the lowland basins land cover changes were more important and generally resulted in higher peak flows and lower summer flows, primarily due to increased runoff. Both land use change and climate effects were important in the upland basins. Climate change had the largest impact in the transitional zones and resulted in higher winter flows, earlier spring peak flows, and lower summer flows. A similar modeling study of a basin located in the Portland, OR metropolitan area, using a single climate change simulation combined with a ArcView Soil and Water Assessment Tool, predicted an increase in overall flow, but a decrease in summer baseflow, by 2040 (Franczyk and Chang 2009).

Monitoring trends and predicting potential future alterations in streamflow patterns is important for water resource managers to ensure sufficient supply to meet demand (Snover et al. 2003, Milly et al. 2008). In the Pacific Northwest, summer low flows define the crucial period of water use and availability, and define system yield capacity. Wiley and Palmer utilized a three-stage modeling approach to evaluate the impacts of climate change on the water supply system for Seattle metropolitan region (Wiley and Palmer 2008). They predicted a decline of 6% per decade in July-September reservoir inflows resulting in a loss of available water in the system of approximately 56,000 acre-feet by 2040. Climate-related changes may reduce overall system yield.

Vano et al. (2009) expanded the analysis to include the Everett and Tacoma water supply systems. They predicted decreased summer reservoir inflows and storage for all three systems. System reliability, however, remained relatively strong assuming current demand.

Summer low flows in streams and rivers may be ecologically important. A substantial body of literature describes the potential deleterious impacts of low summer flows on fish survival (see Crozier et al. 2008, Palmer et al. 2009 and references therein). Potential negative biological impacts of low summer flows include high water temperatures, stranding, low dissolved oxygen, crowding, and disease. Although the strength of salmon runs has been shown to be positively and significantly correlated to summer stream flow in Puget Sound rivers, the actual causative mechanism is unclear due to complicated and interrelated variations between flow, temperature, habitat, and other variables (Mathews and Olson 1980). Rand et al. (2006) evaluated the potential effects of reduced flow and increased water temperature on upriver migration of Pacific salmon in the Frasier River. Lower discharge volumes during the migration period increased survival by decreasing energy requirements of the migrating salmon (making it easier to swim upstream) leading to a stronger pre-spawn population. Higher water temperatures, however, have been shown to increase metabolic rates and increase energy requirements. Presumably, within some range, the energetic benefits of decreased flow will compensate for costs from higher temperatures, yielding no net effect.

Scheuerell et al. (2006) used summer stream temperatures, which are predicted to increase with decreased flow, as a negative factor in survival of Chinook salmon in an effort to model salmon survival according to changes in various environmental conditions. Battin et al. (2007) predicted that Chinook salmon spawner capacity was proportional to minimum discharge during the spawning period; reductions in flow would result in reductions in spawning capacity due to habitat limitations. Low flows are also important for juvenile Coho due to space and food limitations, while low flows may be associated with temperature limitations in other areas (Ebersole et al. 2009). Trout survival and growth have been shown to be negatively associated with low stream discharge (Harvey et al. 2006, Berger and Gresswell 2009).

There remains substantial uncertainty in the predicted changes, related not only to climate change, but also to biological response and potential for adaptation among various species, particularly salmonids(Crozier et al. 2008, Schindler et al. 2008). Biological responses are likely to vary according to the specific stream and basin.

Status and trends

Summer 7-day average low flow is the metric chosen to represent low stream flow conditions. It is widely used and not susceptible to temporary upstream flow changes than may affect one-day low flow calculations (Riggs 1985). Annual values for 7-day average low flow were calculated using gauge data from 14 different locations on unregulated rivers within the Puget Sound, in order to evaluate the status and trends of low flows within the region (Table 1). Data from seven rivers indicated a significantly decreasing trend in 7-day average low flow for the time period on record ($p<0.05$). Data from three other rivers indicated decreasing trends in 7-day average low flow, although with a slightly higher degree of statistical uncertainty ($p<0.10$). Four rivers showed no significant trends in annual 7-day average low flow. Notably, no river system showed significantly increasing trends in annual 7-day average low flow. The average change for the rivers with significant trends in annual 7-day average low flow was -4.4% per decade.

Table 1. Average 7-day Low Flow for the time period of record, the annual rate of change of 7-day low flow, and the probability that the trend is significantly different than zero for selected unregulated rivers and streams in the Puget Sound basin.

			7-DAY AVERAGE LOW FLOW		
River	**Data Years**		**Average Low Flow** (CFS)	**Annual Change** (ΔCFS/Year)	**p** (change≠0)
WRIA 1 – Nooksack					
Nooksack USGS 12213100	1966-2009		1020	-6.3±3.6	0.09
WRIA 3/4 – Upper-Lower Skagit and Samish					
Lower Sauk USGS 12189500	1936-2009		1281	-3.5±2.0	0.08
Upper Sauk USGS 12186000	1929-2009		231	-0.6±0.4	0.09
Thunder USGS 12175500	1931-2009		225	-0.2±0.4	0.55
Newhalem USGS 12178100	1962-2009		45	-0.3±0.1	0.03
Samish USGS 12201500	1945-1970 1996-2009		27	0.01±0.04	0.77
WRIA 5 - Stillaguamish					
Stillaguamish USGS 12167000	1929-2009		255	-0.6±0.4	0.18
WRIA 7 – Snohomish					
Skykomish USGS 12134500	1929-2009		655	-2.2±1.1	0.05
WRIA 8 – Cedar/Sammamish					
Cedar USGS 12114500	1947-2009		22	-0.1±0.04	0.002
WRIA 10 – Puyallup/White					
Puyallup USGS 12092000	1957-2009		216	-1.2±0.5	0.03
WRIA 11 - Nisqually					
Nisqually USGS 12082500	1942-2009		263	-0.7±0.5	0.14
WRIA 13 - Deschutes					
Lower Deschutes USGS 12080010	1946-1963 1990-2009		83	-0.4±0.1	0.001
Upper Deschutes USGS 12079000	1950-2009		29	-0.1±0.04	0.0002
WRIA 16 – Skokomish/Dosewalips					
Duckabush USGS 12054000	1939-2009		72	-0.3±0.1	0.04

There were no consistently strong correlations between the annual 7-day average low flow values for the rivers within WRIA 3/4 (Table 2). Calculated annual 7-day average low flow values from Thunder Creek and the Samish River generally correlate weakly with the other rivers within the group used for comparison. Thunder Creek can be classified as a snowmelt-dominated river. The Samish River is a rainfall-dominated river. The other rivers within the group are all transition rivers. It is possible that the different hydrologic regimes partially explain the lack of correlations in low flow.

Table 2. Pearson's correlation coefficient for annual 7-day average low flow for rivers within WRIA 3/4.

	Lower Sauk	Upper Sauk	Thunder	Cascade	Newhalem	Samish
Lower Sauk		0.84	0.54	0.91	0.73	0.34

Upper Sauk	0.44	0.77	0.80	0.49
Thunder		0.72	0.42	-0.11a
Cascade			0.80	0.23a
Newhalem				0.54

Notes: a. Pearson's r not significantly different than 0 (P>0.05)

Uncertainties

The analysis presented above was derived from data in the public domain. The values and trends for 7-day average low flow were calculated from average daily discharge data from fourteen USGS station located in the Puget Sound region (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.

The 7-day low flow values were calculated for the period from June 1 – November 1; this time period was chosen to avoid the potential capture of winter low flows in the snowmelt-dominated river system (e.g., Thunder Creek). Trends were determined by calculating the slope of the annual 7-day low flow versus year using simple linear regression. Significance was determined by applying the Student's t-test to determine the probability of the slope being significantly different than zero (P<0.05).

The significance of the Pearson's correlation coefficient was determined by estimating the probability that the correlation was different than zero based on the value of the correlation and the sample size. A significant correlation does not indicate a strong correlation.

Summary

Analysis of streamflow data revealed decreasing trends in 7-day average low flow values for seven of 14 guaging stations. Among the remaining stations, none showed significant increasing trends. Substantial inter-annual variation in low flow was evident. Annual 7-day average low flows among the river systems in WRIA 3/4 showed no consistent correlation. The weakest correlations were between the snowmelt-dominated (Thunder Creek), the rainfall-dominated (Samish River) and the remaining river systems. Seven-day average low flow could be a useful indicator of changing conditions in these watersheds.

Literature Cited

Barnett, T. P., D. W. Pierce, H. G. Hidalgo, C. Bonfils, B. D. Santer, T. Das, G. Bala, A. W. Wood, T. Nozawa, A. A. Mirin, D. R. Cayan, and M. D. Dettinger. 2008. Human-induced changes in the hydrology of the western United States. Science 319:1080-1083.

Battin, J., M. W. Wiley, M. H. Ruckelshaus, R. N. Palmer, E. Korb, K. K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America 104:6720-6725.

Berger, A. M., and R. E. Gresswell. 2009. Factors influencing coastal cutthroat trout (Oncorhynchus clarkii clarkii) seasonal survival rates: a spatially continuous approach within stream networks. Canadian Journal of Fisheries and Aquatic Sciences 66:613-632.

Burges, S. J., M. S. Wigmosta, and J. M. Meena. 1998. Hydrological Effects of Land-Use Change in a Zero-Order Catchment. Journal of Hydrologic Engineering 3:86-97.

Casola, J. H., L. Cuo, B. Livneh, D. P. Lettenmaier, M. T. Stoelinga, P. W. Mote, and J. M. Wallace. 2009. Assessing the impacts of global warming on snowpack in the Washington Cascades. Journal of Climate 22:2758-2772.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, N. J. Mantua, J. Battin, R. G. Shaw, and R. B. Huey. 2008. Potential responses to climate change in organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1:252-270.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Cuo, L., D. P. Lettenmaier, B. V. Mattheussen, P. Storck, and M. Wiley. 2008. Hydrologic prediction for urban watersheds with the Distributed Hydrology-Soil-Vegetation Model. Hydrological Processes 22:4205-4213.

Ebersole, J. L., M. E. Colvin, P. J. Wigington, S. G. Leibowitz, J. P. Baker, M. R. Church, J. E. Compton, and M. A. Cairns. 2009. Hierarchical Modeling of Late-Summer Weight and Summer Abundance of Juvenile Coho Salmon across a Stream Network. Transactions of the American Fisheries Society 138:1138-1156.

Elsner, M. M., L. Cuo, N. Voisin, J. S. Deems, A. F. Hamlet, J. A. Vano, K. E. Mickelson, S.-Y. Lee, and D. P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Franczyk, J., and H. Chang. 2009. The effects of climate change and urbanization on the runoff of the Rock Creek basin in the Portland metropolitan area, Oregon, USA. Hydrological Processes 23:805-815.

Hamlet, A. F., P. W. Mote, M. P. Clark, and D. P. Lettenmaier. 2005. Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate 18:4545-4561.

Harvey, B. C., R. J. Nakamoto, and J. L. White. 2006. Reduced streamflow lowers dry-season growth of rainbow trout in a small stream. Transactions of the American Fisheries Society 135:998-1005.

Mathews, S. B., and F. W. Olson. 1980. Factors affecting Puget Sound coho (Oncorhynchus kisutch) runs. Canadian Journal of Fisheries and Aquatic Sciences 37:1373-1378.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change - Stationarity is dead: Whither water management? Science 319:573-574.

Mote, P., D. Canning, D. Fluharty, R. Francis, J. Franklin, A. Hamlet, M. Hershman, M. Holmberg, K. G. Ideker, W. Keeton, D. Lettenmaier, R. Leung, N. Mantua, E. Miles, B. Noble, H. Parandvash, D. W. Peterson, A. Snover, and S. Willard. 1999. Impacts of climate variability and change inthe Pacific Northwest. The JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Mote, P. W. 2003. Trends in snow water equivalent in the Pacific Northwest and their climatic causes. Geophysical Research Letters 30.

Mote, P. W. 2006. Climate-driven variability and trends in mountain snowpack in western North America. Journal of Climate 19:6209-6220.

Mote, P. W., A. F. Hamlet, M. P. Clark, and D. P. Lettenmaier. 2005. Declining mountain snowpack in western north America. Bulletin of the American Meteorological Society 86:39-+.

Mote, P. W., E. Parson, A. F. Hamlet, W. S. Keeton, D. Lettenmaier, N. Mantua, E. L. Miles, D. Peterson, D. L. Peterson, R. Slaughter, and A. K. Snover. 2003. Preparing for climatic change: The water, salmon, and forests of the Pacific Northwest. Climatic Change 61:45-88.

Palmer, M. A., D. P. Lettenmaier, N. L. Poff, S. L. Postel, B. Richter, and R. Warner. 2009. Climate Change and River Ecosystems: Protection and Adaptation Options. Environmental Management 44:1053-1068.

Rand, P. S., S. G. Hinch, J. Morrison, M. G. G. Foreman, M. J. MacNutt, J. S. Macdonald, M. C. Healey, A. P. Farrell, and D. A. Higgs. 2006. Effects of river discharge, temperature, and future climates on energetics and mortality of adult migrating Fraser River sockeye salmon. Transactions of the American Fisheries Society 135:655-667.

Riggs, H. C. 1985. Streamflow characteristics. Elsevier, Amsterdam; New York.

Scheuerell, M. D., R. Hilborn, M. H. Ruckelshaus, K. K. Bartz, K. M. Lagueux, A. D. Haas, and K. Rawson. 2006. The Shiraz model: a tool for incorporating anthropogenic effects and fish-habitat relationships in conservation planning. Canadian Journal of Fisheries and Aquatic Sciences 63:1596-1607.

Schindler, D. E., X. Augerot, E. Fleishman, N. J. Mantua, B. Riddell, M. Ruckelshaus, J. Seeb, and M. Webster. 2008. Climate Change, Ecosystem Impacts, and Management for Pacific Salmon. Fisheries 33:502-506.

Simmons, D. L., and R. J. Reynolds. 1982. Effects of urganization on base-flow of selected South-shore streams, Long Island, New York.Water Resources Bulletin 18:797-805.

Snover, A. K., A. F. Hamlet, and D. P. Lettenmaier. 2003. Climate-change scenarios for water planning studies - Pilot applications in the Pacific Northwest. Bulletin of the American Meteorological Society 84:1513-1518.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Vano, J. A., N. Voisin, L. Cuo, A. F. Hamlet, M. M. Elsner, R. N. Palmer, A. Polebitski, and D. P. Lettenmaier. 2009. Climate Change Impacts on Water Management in the Puget Sound Region, Washington, USA. JISAO Climate Impacts Group, University of Washington, Seattle, WA.

Wiley, M. W., and R. N. Palmer. 2008. Estimating the impacts and uncertainty of climate change on a municipal water supply system. Journal of Water Resources Planning and Management-Asce 134:239-246.

Violations of Instream Flow Rules

Background

Human demands for freshwater resources need to be balanced with the ecological needs of river and estuarine systems (Petts 2009). Instream flow rules, which allocate specific flow and timing regimes in rivers and river system, are meant to legally account for the ecological requirements that may have previously been unconsidered. The Washington Department of Ecology (Ecology) and Department of Fish and Wildlife (WDFW) have developed instream flow rules to "protect and preserve instream resources" (Washington State Department of Ecology 2004), that include fish and fish habitats, water quality, wildlife, aesthetics, and recreation. Instream flow rules are developed by a defined scientific methodology (Washington State Department of Ecology 2003). They do not affect established (senior) water rights or withdrawals. They can limit future (junior) surface water withdrawals, or withdrawals from ground water that is in hydraulic continuity with the surface water, in order to protect minimum instream flows. Instream flow rules may also limit maximum withdrawals or establish closures where it has been determined that there is no water available for further appropriations.

Instream flow rules do not affect exempt groundwater withdrawals, including:

- Stockwatering;
- Single or group domestic, up to 5,000 gallons per day;
- Industrial purposes, up to 5,000 gallons per day; and
- Irrigation of up to one-half acre of lawn or non-commercial garden (see Revised Code of Washington [RCW] 90.44.050).

Instream flow rules exist for many of the rivers and streams within the Puget Sound. Table 1 shows a summary of Instream Flow Rules for basins surrounding the Puget Sound by Water Resources Inventory Area (WRIA).

Alterations of the natural flow regime affect river ecosystems by changing physical habitats, including patterns of longitudinal and lateral connectivity, and by altering the natural cues and patterns of biological response, which could adversely affect native species that have evolved in response to historical flow patterns. Alterations could enhance the success of invasive or introduced species in a river system (Bunn and Arthington 2002). Due to the complexity of natural flow regimes, the establishment of simplified instream flow rules based on minimum flow requirements or rules of thumb may not be protective of natural resources; i.e., it is not clear whether instream flow rules are protective of native flora and fauna (Arthington et al. 2006, Naiman et al. 2008). Several studies have suggested the adoption of flow rules and management targets that are more considerate of all aspects of the natural flow regime (Bunn and Arthington 2002, Arthington et al. 2006, Naiman et al. 2008, Petts 2009, Poff et al. 2010).

A measure of the management effectiveness of freshwater resources is to compare actual instream flows with the instream flow rules. A high percentage of instream flow rule violations could indicate an over-allocation of freshwater in a basin. An increasing trend in violations could indicate that the freshwater demands are increasing. For the purposes of this report, violations were determined by comparing the instream flow rules to the average daily flow at specified gauging stations. A violation was noted when the average daily flow was less than that specified in the instream flow rule. The average percent of violation days per month were calculated for the time period of the instream flow rule. Trends were evaluated for the period from October to June or during the typically water-critical period from July toSeptember (see Table 2). Trends were determined by simple linear regression over time; trends significantly different that zero (P<0.05) were noted..

Violations for instream flow rules were calculated for eight rivers, with the intent of evaluating at least one river or stream from each of the WRIAs in the Puget Sound watershed. The selection of rivers is shown in Table 1.

Table 1. Summary of Instream Flow Rules for Water Resource Inventory Areas (WRIA) surrounding the Puget Sound.

Water Resources Inventory Area	Instream Flow Rule	Date	Closures
WRIA 1 - Nooksack	173-501 WAC	12/4/85	Yes
WRIA 2 - San Juan	No		
WRIA 3/4 - Lower Skagit-Samish and Upper Skagit	173-503 WAC ,	4/14/01, Update 6/15/06	No
WRIA 5 - Stillaguamish	173-505 WAC	9/26/05	Yes
WRIA 6 - Island	No		
WRIA 7 - Snohomish	173-507 WAC	9/6/79	Yes
WRIA 8 - Cedar-Sammamish	173-508 WAC	9/6/79	Yes
WRIA 9 - Duwamish-Green	173-509 WAC	6/6/80	Yes
WRIA 10 - Puyallup-White	173-510 WAC	3/21/80	Yes
WRIA 11 - Nisqually	173-511 WAC	2/2/81	Yes
WRIA 12 - Chambers-Clover	173-512 WAC	12/12/79	Yes
WRIA 13 - Deschutes	173-513 WAC	6/24/80	Yes
WRIA 14a - Kennedy-Goldsborough	173-514 WAC	1/23/84	Yes
WRIA 15 - Kitsap	173-515 WAC	7/24/81	Yes
WRIA 16/14b - Skokomish-Dosewalips	No		
WRIA 17 - Quilcene-Snow	173-517 WAC	12/31/09	Yes
WRIA 18 - Elwha-Dungeness	No		

Status and Trends

None of the river systems evaluated consistently met the instream flow rules (Table 2). In five of the eight river systems, there were at least two months per year when actual flows did not meet the instream flow requirements at least 50% of the time. Flows in the Stillaguamish River failed to meet instream flow rule requirements 90% of the time during the July-August-September period. This is the highest percent of violation of any river evaluated.

Table 2. Summary of percent violations of Instream Flow Rule for selected rivers in the Puget Sound. Period is effective dates of Instream Flow Rule. Violations occurred when average daily flow at gauging station was less than value specified by Instream Flow Rule. Overall average for the time period and annual percent change are shown. Water Resources Inventory Areas that are not shown do not have established Instream Flow Rules.

Water Resources Inventory Area	USGS Gauge Station ID	Period	Average (Trend per year); %Violations per year[9]	
			Oct – June	July-Aug-Sept
WRIA 1 - Nooksack	USGS 12213100 Nooksack River at Ferndale, WA	1986-2009	34 (- 0.3)	72 (- 0.3)
WRIA 3/4 - Skagit	USGS 12200500 Skagit River near Mount Vernon, WA	2002-2009	24 (+ 1.8)	54 (- 0.7)
WRIA 5 - Stillaguamish	USGS 12167000 NF Stillaguamish River near Arlington, WA	2005-2009	20 (- 1.5)	90 (- 2.5)
WRIA 7 - Snohomish	USGS 12144500 Snoqualmie River near Snoqualmie, WA	1979-2009	23 (+ 0.1)	53 (+ 1.3)
WRIA 8 - Cedar-Sammamish	USGS 12119000 Cedar River at Renton, WA	1979-2009	16 (- 0.3)	21 (- 0.9)
WRIA 9 - Duwamish-Green	USGS 12113000 Green River near Auburn, WA	1980-2009	15 (- 0.3)	55 (- 0.3)
WRIA 10 - Puyallup-White	USGS 12101500 Puyallup River at Puyallup, WA	1980-2009	7 (- 0.3)	10 (- 0.6)
WRIA 11 - Nisqually	USGS 12082500 Nisqually River near National, WA	1981-2009	23 (- 0.3)	21 (- 0.0)
WRIA 12 - Chambers-Clover[10]				
WRIA 13 - Deschutes[11]				
WRIA 14a - Kennedy-Goldsborough[12]				
WRIA 15 - Kitsap[13]				
WRIA 17 - Quilcene-Snow[14]				

Generally, the highest percent of violation of instream flow rules occurred in August and September . There were no significant trends of the percent violations of the instream flow rule over time for any of the river systems evaluated (P>0.05).

Uncertainties

This analysis uses average daily discharge data from the eight USGS stations specified in Table 2 (United States Geological Survey 2010b). The datasets include qualification codes indicating whether data are provisional or have been approved (United States Geological Survey 2010a). We avoided using provisional data in this analysis, and we omitted data from gauging stations for which advisory notes warning against unreliable data quality had been posted.The gauging stations on the NF Stillaguamish River near Arlington (USGS 12082500) and the Nisqually

River near National (USGS 12082500) advised of poor data quality during storms or high flow conditions. High flow conditions would not result in violations of the instream flow rules and so this did not affect the analysis.

The development and application of Instream Flow Rules is relatively recent (see Table 1). Consequently, most stations offer only a limited number of years from which to evaluate data. The relatively short time period and high interannual variability precluded detection of significant long term trends.

Summary

All streams showed violations of the instream flow rules, most commonly occurring in August and September. Notably, flow levels in the Stillaguamish River were below instream flow requirements approximately 90% of the time during the summer months. The Puyallup River exhibited the lowest percent of instream flow rule violations of any river evaluated. The monthly average percent violations did not exceed 25% for any month of the water year.

Literature Cited

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications 16:1311-1318.

Bunn, S. E. and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30:492-507.

Naiman, R. J., J. J. Latterell, N. E. Pettit, and J. D. Olden. 2008. Flow variability and the biophysical vitality of river systems. Comptes Rendus Geoscience 340:629-643.

Petts, G. E. 2009. Instream Flow Science For Sustainable River Management. Journal of the American Water Resources Association 45:1071-1086.

Poff, N. L., B. D. Richter, A. H. Arthington, S. E. Bunn, R. J. Naiman, E. Kendy, M. Acreman, C. Apse, B. P. Bledsoe, M. C. Freeman, J. Henriksen, R. B. Jacobson, J. G. Kennen, D. M. Merritt, J. H. O'Keeffe, J. D. Olden, K. Rogers, R. E. Tharme, and A. Warner. 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology 55:147-170.

United States Geological Survey. 2010a. Provisional data for Washington.

United States Geological Survey. 2010b. USGS Washington Water Science Center.

Washington State Department of Ecology. 2003. A Guide to Instream Flow Setting in Washington State. Water Resources Program, Olympia, WA.

Washington State Department of Ecology. 2004. Setting instream flows and allocating water for future out-of-stream uses. Olympia, WA.

Chapter 2B. The Socio-economic Condition of Puget Sound (outline)

Section 1. Outline

The current presentation of Section 2B is in an outline form—a more complete section will posted later. The objectives of Section 2B are: (a) to establish a framework for organizing existing and emerging information from the breadth of the social sciences - what we will refer to as the "human dimensions' of Puget Sound restoration and preservation, and (b) to review existing and potentially high value human dimension data that can be used as indicators for the social and economic "state" of the region, and the "governance" of the Puget Sound marine resources. Outline of Section 2B

1. Approach

a. Distinction between State and Governance indicators. Data about human populations, economic activity, land use practices, and other 'state' indicators can paint a picture of the status of human well-being in the Puget Sound region. "Governance" information describes the characteristics of decision-making processes implicated in Puget Sound recovery.

b. Analytical frameworks for encompassing state and governance indicators in regional ecosystem planning. Using well-documented frameworks such as the Common Pool Resource and Institutional Analysis and Development approaches as organizing schemes.

c. Communicating levels of scientific understanding and confidence in assessments of different elements of human well-being.

The State of the Puget Sound Region: PSP Action Agenda in a Regional Context

a. Restoration Goals/Objectives in PSP Action Agenda

b. Relevance of the general regional "state" of human well-being and its elements to the PSP Action Agenda. Understanding the human and geographic context of Puget Sound restoration is important because opportunities and threats to restoration are affected by property use in the region; and in turn, restoration and protection actions (or their absence) impact the surrounding population.

c. Demography and Land Use -describe the state of the built environment and human geography, including population distribution, migration sources and patterns, and existing land-uses and regulations in place.

d. Population Well-being - focus on general state measures of human health and well-being. Include distributional equity: the social distribution of the environmental goods and services, including human health risks resulting from the Puget Sound marine system.

e. Economic structure of the Puget Sound –describe the state of the region's economy; evaluate approaches to disaggregating marine-specific and more generally, economic activities dependent on the state of natural resources in the region.

Land Use and Governance

a. Review what is known about the governance of land use allocation and distribution decisions.

b. Review governance of marine waters and tide lands. Include what is known about the capacity for responsible institutions to make fair and competent natural resource allocation and distribution decisions.

Stewardship Capacity. Stewardship is the capacity to act, enabled by available human, social, cultural and economic capital.

a. Inventory the human, social, financial, built, and cultural capital that can enable stewardship of the Puget Sound marine system.

Chapter 3. Impacts of Natural Events and Human Activities on the Ecosystem

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Section 1. Introduction

The health of the Salish Sea ecosystem is directly influenced by both human activities and natural events (Ruckelshaus and McClure 2007). The mechanisms through which these actions lead to ecosystem changes are complex. Yet, the identification of threats and their myriad impacts is necessary to strategically and effectively manage their causes and their impacts on the Salish Sea (strategies discussed in Chapter 4 of the Puget Sound Science Update). In this Chapter, we identify threats to the Salish Sea ecosystem and provide empirical evidence for the causal linkages between high priority threats and their associated ecosystem impacts. Although we do not comprehensively or systematically attempt to identify and propose indicators that will allow us to track change in the health of the Salish Sea (see Chapter 1), we do identify potential indicators from the literature which together with the information we reviewed can serve as a basis for selecting indicators of the health of Puget Sound.

The goals of this Chapter are to:

1. Identify terrestrial, freshwater and marine derived threats to the Salish Sea ecosystem including the freshwater and terrestrial environments.
2. Review threat ranking schemes and identify the threats with highest impact.
3. Use a conceptual model to examine the causal relationships between threats and their impacts on the environment (Driver-Pressure-State-Impact-Response (DPSIR) framework). We emphasize what is known about the geographic scope, severity, irreversibility, imminence, and uncertainty of high impact threats and identify associated knowledge gaps.
4. Identify ecosystem models that have been developed for the Puget Sound region that identify and rank ecosystem threats or that help us identify indicators of ecosystem health.

1. Methods

We conducted a literature review to synthesize information on threat ranking schemes, threats described as having the greatest ecological impact on the Salish Sea ecosystem, the impacts of individual threats, and models to use as tools to evaluate the impacts of human activities. We report on peer-reviewed scientific literature but also include relevant technical memos and government reports when appropriate. We do not include original analyses. Therefore, our Chapter serves to synthetically report on what is already known in the scientific literature and identify knowledge gaps.

We recognize that human activities that threaten ecosystems may also contribute positively to human health and wellbeing. For example, shoreline hardening has negative physical and biological impacts such as contributing to the loss of beaches and spawning habitat for fish but also has positive impacts on human wellbeing by preventing erosion and loss of property at local scales (see shorelines in this Chapter). In this first edition of the Chapter on threats, we focus on the negative ecological and physical impacts of human activities. In future editions of this Chapter, we anticipate both a review and evaluation of the threats relative to human systems (economies, human health and wellbeing, cultural resources, etc.) as well as ecological systems.

More specifically, future editions should include evaluations of the linkages between threats, human systems and ecological systems, highlighting not simply how enhancement of one system is costly to the other, but how the two systems benefit from each other. Future integration of the two systems and is necessary as a foundation for analyzing tradeoffs associated with various conservation actions.

Next Step: Impacts of threats to human health and wellbeing – positive or negative - were not addressed in this chapter and should be included in future editions.

Causal relationships between threats and their impacts to the environment

To help us better understand the high impact threats and their effects on the ecosystem, we use a conceptual framework designed to examine the relationship between human activities and the environment, namely "Driver-Pressure-State-Impact-Response" (DPSIR) (e.g., Langmead et al. 2009, Carr et al. 2007, Elliot 2002). Drivers are factors that result in pressures that, in turn, cause changes in the ecosystem. Drivers are both natural (e.g., natural climate variability, earthquakes, tsunamis) and anthropogenic (e.g., residential and urban development, human-caused climate change). In principle, human drivers can be changed via responses such as regulation, restoration, and education and natural environmental drivers cannot be controlled but must be accounted for when assessing interactions among drivers and pressures or the effectiveness of management responses. Pressures are factors that cause changes in a state or condition and are caused by specific drivers. For example, the driver "residential, urban and industrial development" can cause the ecological pressures of pollution and vegetation loss. State variables describe the condition (including physical, chemical, and biotic factors) of the ecosystem such as the presence of 6 parts per million of a given contaminant in Commencement Bay. Impacts comprise measures of the effect of change in these state variables such as loss of biodiversity, declines in productivity and yield, etc. Responses are the actions (regulatory, management or educational activities) that are taken in reduce the pressures and impacts caused by various drivers in order to achieve a desired state (e.g., cleaner water).

A DPSIR approach allows us to organize and present a wide range of issues that shape our understanding of the problem: the contribution of human activities to the problem, the extent and magnitude of the problem and the harm it causes in the ecosystem, and the range of possible strategies we might employ to mitigate it. These three ideas are reflected in three Chapters, collectively, of the Puget Sound Science Update. Using DPSIR terminology, the present Chapter (3, Impacts of Natural Events and Human Activities on the Ecosystem) discusses the Threats, or Drivers and Pressures in the system (and associated states and impacts); Chapter 2, Biophysical status of Puget Sound reviews the status of and trends in current condition of the ecosystem (State and Impacts); Chapter 4, Effectiveness of Strategies to Protect and Restore the System addresses the human Response to the problem.

We use the DPSIR framework because it: (1) identifies likely causal linkages between human activities and changes in ecosystem states; (2) simplifies the complex relationship between human activities and changes in the environment; (3) is a tool for communicating complex relationships and potential solutions between policy makers, scientists and the general public; (4) provides a framework for identifying indicators and what they indicate (e.g. indicators of pressures and states); (5) allows for a better understanding of the likely effects of response actions on the desired state; and (6) is widely used in the peer–reviewed literature. This framework has been used to identify indicators (Mangi 2007), to identify issues associated with pollinator loss (Kuldna et al. 2009), to address wind power management (Elliot 2009), to model choices associated with ecosystem recovery (Langmead et al. 2009), to create a conceptual framework for marine protected areas (Ojeda-Martinez et al. 2009), and for socio-ecological modeling (Langmead et al. 2009). The DPSIR framework can be modified to examine the effects of various drivers on human health and wellbeing.

Method used: To help us better understand the high impact threats and their effects on the ecosystem, we use a conceptual framework designed to examine the relationship between human activities and the environment, namely "Driver-Pressure-State-Impact-Response."

In the following text, we treat most of the identified high impact threats as "drivers" or "pressures" and examine their impact on ecosystem states.

Results

Threat Ranking and identification of high impact threats

Our literature review revealed qualitative approaches for identifying and ranking the relative impacts of threats to the Salish Sea ecosystem. Of the six sources that identified threats to this ecosystem, all of them were governmental efforts, and one of ranked threats using an expert based approach (Table 1):

- "Identification, Definition and Rating of Threats to the Recovery of Puget Sound," (Neuman et al. 2009). This technical memo describes an expert review process using the Miradi Open Standards for the Practice of Conservation to identify threats to this ecosystem and rank threats as "low", "medium", "high", and "very high".
- Puget Sound Partnership Action Agenda (2008, see Appendix, Threats and Drivers Summary in the Appendix). A categorized list of threats impacting Puget Sound that was developed as part of a DPSIR Demonstration Project in support of PSP's Action Agenda.
- Washington Department of Ecology list of significant threats to Puget Sound (WA DoE 2010): This list of threats emphasizes the agency's focus on air and water contaminants.
- U.S. Environmental Protection Agency Region 10 indicators for Puget Sound/Georgia Basin (US EPA 2010): Not a list of threats per se, but rather a list of broad-level indicators that are proposed or currently being monitored. Some of these indicators reflect one or more threats.
- Washington Biodiversity Council (WBC 2007): Identification of threats associated with the loss of biodiversity but also relevant to ecosystem processes in general.
- Significant threats to nearshore habitats in Cherry Point, WA (Hayes and Landis 2004): list of threats and impacts identified through an environmental assessment conducted in 2001 using a Regional Risk Assessment model to characterize high versus moderate risk threats. Comprising a small sub-region within the Puget Sound/Georgia Basin system, the Cherry Point assessment provides a useful illustration of how the specific scale of assessment can affect the level of risk posed by a particular threat.

We also include two threat identification and ranking processes that resulted from a similar restoration effort in the Chesapeake Bay and for the California Current ecosystem:

- Chesapeake Bay Program's lists of pressures (CBP 2010): despite the differences between the Puget Sound and Chesapeake Bay (in terms of climate, structure, etc.), they appear to share many of the same threats. As with EPA's list above, Chesapeake Bay Program's list includes a collection of indicators for monitoring the status and impacts of important threats.
- Human impacts to the California Current marine coastal ecosystems (Halpern et al. 2009). This study maps the cumulative human impacts to California Current marine ecosystems. This system has direct physical and biotic linkages to the Salish Sea and this analysis included the Salish Sea. This assessment reflects differences in relative impacts of various threats as a function of scale (as well as important biophysical differences between marine coastal and inland estuarine systems).

Identifying the most important threats: Is one threat more important than the other in Puget Sound? Threats in Puget Sound were ranked as low, medium and high based on expert opinion in various venues (Table 1). Our review of the literature suggests that threat identification and ranking approaches used in the Puget Sound region largely lack peer-review and are not necessarily comprehensive, indicating the need for a more quantitative and systematic approach that addresses uncertainty surrounding the relative magnitude of threats. We propose approaches to get to the answer in our introduction and model sections.

Table 1. Comparison of Threat Identification and Ranking Lists for Puget Sound/Georgia Basin and Comparable Ecosystems1. Although there is considerable overlap in the threats identified in each scheme presented, each presents a unique threat list. X's indicate threats that were identified as significant but were not ranked according to their relative importance.

PSSU Threats	PSP Open Standards Rating	Washington Department of Ecology	EPA Region 10	Washington Biodiversity Council	Cherry Pt., WA	CA Current Coastal Assessment	Chesapeake Bay Program
Residential, Commercial, & Industrial Development	Very High	X	X	X	High		X
Climate Change	Very High			X			X
Non-native & Invasive Species	High		X	X	Medium	Medium	X
Point & Non-point Water Pollution	High	X	X	X	Medium	Medium	X
Shoreline Modification	High	X		X	Medium	Medium	
Species Harvesting	High		X	X			X
Transportation	High			X			X
Air Pollution & Atmospheric Deposition	Medium		X			High	X
Forest Practices	Medium		X	X			X
Oil & Hazardous Spills	Medium	X	X	X	Medium	Medium	X
Recreational Activities	Medium			X	High	Medium	
Water Demands, Withdrawals & Diversions	Medium			X			X
Agriculture Practices	Low			X	High		
Aquaculture Practices	Low		X			Medium	X
Derelict Gear & Vessels	Low					Medium	
Dredging Activities	Low						
Physical Disturbance/Disruption of Species	Low				High	Medium	
Military Exercises	Low						
Mines	Low						

There is fairly high consistency among ranking schemes in the threats identified for Puget Sound and for similar ecosystems (Table 1). Because only one assessment effort ranks threats Sound-wide (Neuman et al. 2009), we use this scheme to help us focus our efforts on the threats thought to have the greatest impact on the health of Puget Sound. Specifically, we only review the "very high" and "high" threats identified in Neuman et al. (2009). We did not have time to address one of the "High" ranked threats, unsustainable harvest, and recommend that future editions of this Chapter describe this threat (Table 2). To better fit into our DPSIR approach, we characterize the threats slightly differently from Neuman et al. (2009). For example, "Physical disturbance/disruption to species" which is a "Low" rank threat under Neuman et al. (2009) is partially covered as a "State" under our more comprehensive driver, "Residential, Commercial and Industrial Development". The use of DPSIR and resulting change in threat categories resulted in partial reviews of some lower ranking threats. For the threats reviewed, our work is incomplete and we welcome input from experts to help make this product more comprehensive. To help with this process, we highlight obvious gaps in our assessment with placeholders in the text.

Table 2. Threats and their ranks from Neuman et al. (2009) reviewed in this Chapter.

Threats from Open Standards Neuman et al (2009)	Open Standards ranking	Where we include threats from the Open Standard process in this chapter
Climate Change	Very High	Climate Change
Residential, Commercial, Port & Shipyard Development	Very High	Residential, Commercial and Industrial Development
Surface Water Loading and Runoff from the Built Environment	High	Residential, Commercial and Industrial Development
Roads,Transportation and Utility Infrastructure	High	Residential, Commercial and Industrial Development
Shoreline Armoring	High	Shoreline Modification
Dams, Levees and Tidegates	High	Shoreline Modification
Invasive Species (marine, freshwater and terrestrial)	High	Invasive and Non-native Species
Point & Non-point Water Pollution	High	Pollution – focus on impacts to biota
Unsustainable Species Harvest	High	Not covered – high priority for next update
Air Pollution & Atmospheric Deposition	Medium	Pollution - incomplete
Forest practices	Medium	Not covered
Oil & Hazardous Spills	Medium	Pollution - incomplete
Recreational activities	Medium	Not covered
Water Demand, Withdrawals and Diversions	Medium	Residential, Commercial and Industrial Development – incomplete
Agriculture practices	Low	Not covered
Aquaculture practices	Low	Not covered
Derelict Gear & Vessels	Low	Not covered
Dredging activities	Low	Not covered
Physical disturbance/disruption to species	Low	Residential, Commercial and Industrial Development – just terrestrial
Military Exercises	Low	Not covered
Mines	Low	Not covered

Of the high impact threats identified by the Puget Sound Partnership, we addressed: Climate Change, Residential, Commercial and Industrial Development, Shoreline Modification, Invasive and Non-native Species, Pollution, with a focus on impacts to organisms

High impact threats not addressed in this chapter: Unsustainable species harvest

We did not review all threats identified by the Open Standards process (Table 1).

Information Needs

Identifying the most significant threats and the most important management strategies is extremely complex, especially if threats interact and their effects are multiplicative in nature. For example, threats to marine ecosystems often include simultaneously terrestrial, freshwater and marine based effects (Halpern et al. 2007). The current approaches to threat identification and ranking described above largely lacks peer-review. Our review of the literature suggests the need for a more comprehensive, quantitative and systematic assessment that addresses uncertainty surrounding the relative magnitude of threats. There are many approaches to both identifying and ranking threats in the published literature (e.g., Iannuzzi et al. 2009, Newsome et al. 2009, Selkoe et al. 2008, Halpern et al. 2007, Given and Norton 1993). Our review revealed the following considerations when conducting such a threat assessment:

- Importance of identifying clear objectives that define the 1) geographic scope, 2) ecosystem(s), ecological communities, and species of interest (what is threatened), and the 3) temporal scope.
- A systematic and comprehensive assessment of threats (e.g., Halpern et al. 2007).
- Expert opinion is often used in the absence of models for identifying and ranking threats. The literature suggests the following considerations when using this approach:
    - Quantitatively assess vulnerability and mathematically embrace uncertainty in our knowledge about the threats and their associated impacts when developing threat ranks (e.g., Cooke and Goossens 2004). Consistency between the top threats volunteered by experts and the top threats revealed using vulnerability scores from these same experts can be low (Halpern 2007, Teck et al. 2010, Payne et al. 1992; Lichtenstein and Slovic 2006) and suggests the importance of a more quantitative approach. There are many approaches for addressing uncertainty (e.g., Teck 2010, Iannuzzi et al., 2009, Halpern et al. 2007, Garthwaite et al. 2005, Cooke and Goossens 2004, Morgan 2003, Cleaves 1994) and we suspect that a Bayesian belief network approach (e.g., Garthwaite et al. 2005) could also be applied if threats and their consequences can be expressed as probabilities and as discrete values.
    - Recommend developing criteria when selecting experts to make sure that the appropriate representation and level of knowledge is included (e.g., level of education, type of research experience, management experience, type of organization, etc.)
    - Address sources of bias: (1) self interest or personal values of those included as experts (see Cleaves 1994); (2) institutional, educational and sex biases (see Halpern 2007 for an analytical approach for addressing this issue)
    - Integrate published material and expert opinion (e.g., information on magnitude, extent and uncertainty associated with threats; Iannuzzi et al., 2009).
    - Integrate expert based threat ranking with quantitative information (e.g.,Teck et al. 2010 Iannuzzi et al., 2009) to provide a systematic foundation for ecosystem-based management
- There are modeling approaches that help both identify and rank threats and are discussed in the Modeling section and the concluding paragraphs of this Chapter of the Update.

Next Step: Work is needed to more comprehensively evaluate the impact of single threats as well as the interactions among them. We included placeholders to guide future editions of this section.

Key information gap: Quantitative and analytical approaches to ranking threats in Puget Sound

Driver: Climate Change in the Salish Sea Ecosystem

Whereas weather is the daily to seasonal changes in patterns of temperature, precipitation, humidity, and wind; Climate change is the long term trend of these patterns. Some short-term climate variation is normal from cycles of the Pacific Decadal Oscillation and El Niño-Southern Oscillation; however, natural causes and natural variability alone cannot explain the rapid increase in global temperatures in the last 50 years (Climate Impacts Group 2009). Although these natural cycles complicate determining the full extent and cause of increased temperatures, most evidence confirms that at least some of the rise in temperature is attributable to the buildup of greenhouse gases (Hegerl et al. 2006). The average net effect of global human activities since 1750 has been one of warming, with a radiative forcing of +1.6 [+0.6 to +2.4] W/m2 (IPCC 2007). In comparison, changes in natural solar irradiance since 1750 are estimated to have caused a relatively small radiative forcing of +0.12 [+0.06 to +0.30] W/m2 (IPCC 2007). This range of natural and human factors driving the warming or cooling influences on global climate plays an essential role in shaping ecosystems.

A conceptual model such as Driver-Pressure-State-Impacts-Response (DPSIR) can provide context for the climate change threat (Figure 1). In the following sections, we use DPSIR terminology to help evaluate climate change related pressures to the ecosystem in terms of the classes of processes that often affect the structure (state) and function (impact) of the ecosystem. The strategies to mitigate and adapt to climate change are discussed in more detail in Chapter 4.

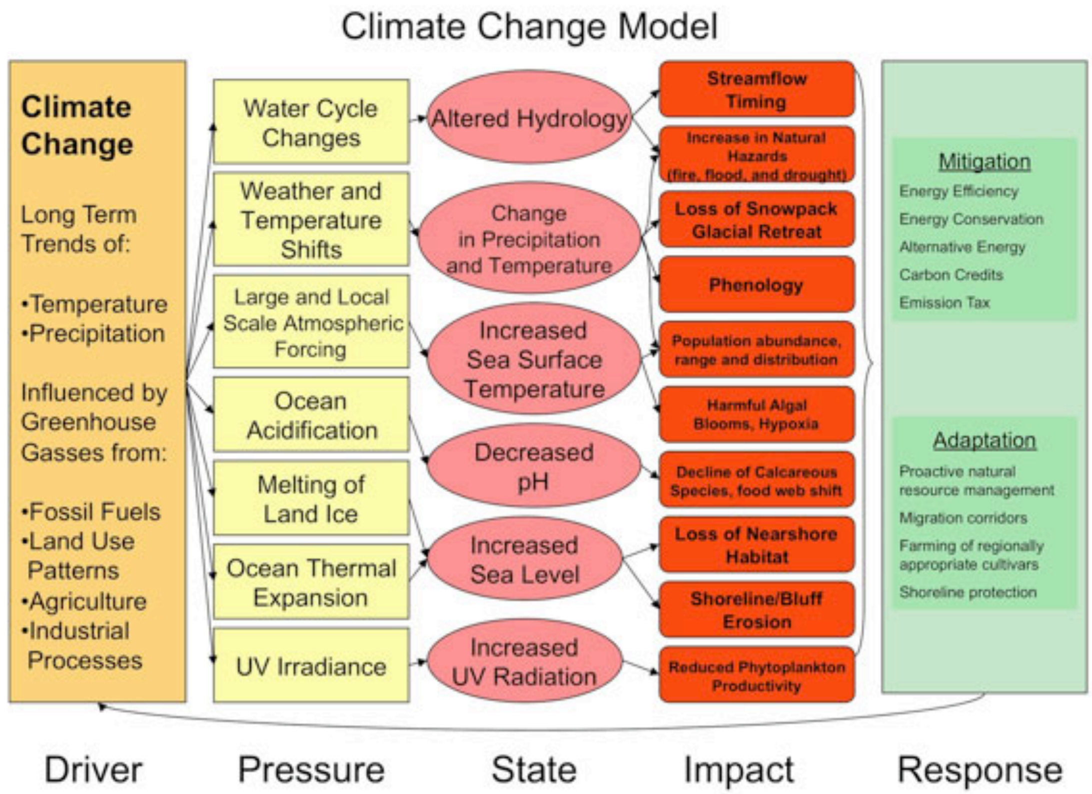

Figure 1. Driver-Pressure-State-Impacts-Response conceptual model for Climate Change.

The pressures that climate change exerts on the Salish Sea ecosystem fall into six general classes of processes that affect its structure and functioning: (1) water cycle changes; (2) weather/temperature change, (3) ocean thermal expansion/melting of land ice;(4) large and local scale atmospheric forcing; (5) ocean acidification; and (6) ultraviolet irradiance. Each in turn contributes to changes in ecosystem states. Note, however, that more than one pressure can contribute to a given state change; similarly, many system-level impacts are driven by multiple state changes. These various relationships are reviewed and described in greater detail below.

Although not explicitly addressed in this iteration of the Puget Sound Science Update, the impacts of climate change on the citizens of the Salish Sea ecosystem are important to consider in addition to impacts on the ecosystem. A changing climate will affect many facets, including impacts to infrastructure and human health and wellbeing. These impacts should be included in future updates.

1. Pressure: Water Cycle Changes

Hydrology in the Salish Sea ecosystem is governed by three watershed regimes: (1) high elevation is snowmelt dominant, (2) mid-elevation is transient with a rain/snow mix, and (3) low-elevation is rain dominant. Transient watersheds are most prevalent (Climate Impacts Group

2009) and will be the first to show a quantifiable response to changing climate as it changes to a rain dominant regime associated with increased temperature.

State: Altered Hydrology

Timing of peak streamflow varies seasonally between the three watershed regimes (Climate Impacts Group 2009). Snowmelt dominant streamflow peaks when temperatures begin to rise and melt the snowpack during May-July. Rain dominant streamflow peaks when precipitation peaks, typically during November-January. Transient streamflow is unique in that it peaks twice; once during November-January coinciding with peak precipitation and again during May-July coinciding with the snowpack melting.

Impacts:

The watersheds with streamflow based wholly or partially on snowmelt are predicted to have the greatest hydrological shifts associated with climate change. Snowmelt plays an integral role in the seasonal timing of streamflow and thus affects the region's water supply. Impacts to the water cycle are likely to include earlier peak stream flows, decreasing runoff in spring/summer, and increasing runoff in autumn/winter.

In watersheds with snowpack, especially the transient watersheds, winter and spring warming are likely to cause a cascade of events that lead to increased snowpack melt. Warming lessens the snow-to-precipitation ratio, resulting in reduced snowpack which in turn further increases the absorption of solar radiation by the land surface, triggering the snow albedo feedback to increase the rate of melting (Mote et al 2008a). This chain of events is responsible for the advanced timing of streamflow (Hidalgo 2009).

The shift in the timing of streamflow of snowmelt-dominant basins has been evident since the late 1940's (Stewart et al 2005). In recent decades throughout the western US, streamflow timing has occurred one to four weeks earlier than in the 1950's through the mid-1970's, with trends being strongest for mid-elevation transient zones; the timing of advance in streamflow timing is significant ($p = 0.05$) (Hidalgo 2009).

This advance in streamflow timing of basins with snowpack was shown with three related measures:

1. The center timing of streamflow, which is the average day by which time half of the annual streamflow has passed, occurs earlier in the spring. The early shift is present throughout western North America, including the Salish Sea region, for the period of 1948–2002 (Stewart et al 2005). For the Puget Sound Basin specifically, it is projected that center timing will occur between 2-5 weeks earlier during the 2020's than it did from 1917-2006 (Climate Impacts Group 2009).
2. An advance of timing in the snowmelt onset, with mid-elevations having the largest advance due to being more sensitive to early temperature changes than high-elevations. Overall, from 1948-2002, spring pulse onset occurred 10-30 days earlier in western North America (Stewart et al 2005).

3. Decreased spring and early summer fractional flows and increasing fractions of annual flow occurring earlier in the water year (Stewart et al 2005). In the Puget Sound Basin, flows for 2006 conditions were higher in late winter and early spring, and lower in late spring and summer compared to 1915 levels, which reflects the generally warmer winter temperatures for 2006 (Cuo et al 2009). The trend of winter peaks becoming higher and summer peaks becoming lower is projected to continue throughout the 21st century (Climate Impacts Group 2009). Transient watersheds in particular will see the largest shift in interseasonal distribution of streamflow. As snowpack decreases, streamflow is projected to shift from having a doublepeak, to only a single peak in December, associated with the loss of snowmelt and increasingly rain-dominant behavior.

In contrast to snowmelt dominated basins, rain dominant watersheds are relatively unaffected by climate change (Stewart et al 2005). The center timing of low elevation rain-dominant watershed basins are trending in the opposite direction of high-elevation basins. Streamflow center timing of rain-dominant streams occurred 5-25 days later in 2002 compared to historical values from 1948-2002 (Stewart et al 2005). This suggests that the trends seen in high-elevation basins are most likely attributable to temperature changes, rather than precipitation (Adam et al 2009; Climate Impacts Group 2009). In addition, mean annual streamflow has remained constant over the past 50 years despite seasonal shifts (Cuo et al 2009; Stewart et al 2005).

Pressure: Weather/Temperature Shifts

In the Pacific Northwest, regional climate models generate shifts in snow cover, cloud-cover, and circulation patterns associated with interactions between large-scale climate change and the regional topography and land–water contrasts (Salathe et al 2008). Changes in weather conditions alter the state of temperature and precipitation trends over the region. A majority of impacts to the system within the Salish Sea ecosystem are a result of interactions between increased temperature and precipitation pattern shift.

State: Increased Temperature

According to the IPCC Fourth Assessment Report (2007), globally the earth's climate rose 0.7°C over the last century. Over this same time period, the temperature in the Puget Sound basin slightly exceeded the global average, generally registering a 0.8°C increase (Mote 2003). This rise in temperature is not expected to level off anytime soon, in fact the rate of change is predicted to increase over the coming century. The Climate Impacts Group predict average temperature rise in the Pacific Northwest of 1.1°C (range of projections from all models: +0.6°C to +1.8°C) by the 2020s; 1.8°C (range: +0.8°C to +2.9°C) by the 2040s; and 2.9°C (range: +1.6°C to +5.4°C) by the 2080s compared to 1970-1999 temperatures. Warming is expected to occur throughout all seasons with the largest increase found in the summer months (Climate Impacts Group 2009).

State: Precipitation Pattern Shifts

Throughout North America, precipitation on average has increased over the last century (Field et al 2007) with precipitation in the Pacific Northwest exceeding the global average by 13%- 38%

(Mote 2003). Southern British Columbia also had an increase in precipitation between 5-35% over the 20th century (Zhang et al 2000). Agreement among models on estimations of future amounts of precipitation in the Pacific Northwest is lacking. When results among these models are averaged, the overall projected changes in our region are modest, with a 1.3% precipitation increase; range of projections from all models: (-9 to +12%) by the 2020s; +2.3% (range: -11 to +12%) by the 2040s; and +3.8% (range: -10 to +20%) by the 2080s compared to 1970-1999 precipitation levels (Climate Impacts Group 2009).

Most models agree that there will be large seasonal changes, especially toward wetter autumns and drier summers (Climate Impacts Group 2009; Jakob & Lambert 2009). The regional models also predict increases in extreme high precipitation over the next half-century, especially in the Puget Sound area (Climate Impacts Group 2009). The seasonality, frequency and intensity of extreme events, including storms, must be considered in addition to the annual amount of precipitation since extreme events cause immediate damage to the ecosystem, versus a gradual shift in conditions over years.

Impacts:

Snowpack and Glaciers

The impact of rising temperature with the most far reaching effects is the loss of snowpack and glacial retreat. Regardless of how much precipitation falls in our region, ambient air temperature determines how much of that falls as snow or rain. Increased temperatures reduce the length of the snow season and increase the elevation of snowline, and thus decrease the amount of snowpack in Puget Sound.

Snow water equivalent is a common measurement of snowpack. It is the amount of water held within the snowpack and can be thought of as the depth of water that would occur if the entire snowpack melted. Stoelinga et al (2009) determined snow water equivalent declined in the Cascade Range by 23% (95% CI: ±28%) from 1930-2007. During that same time period, the Washington State portion of the Cascade Range may have had a slightly higher percent loss, ranging from 15-35% with rising temperatures being the main source for the decline (Mote et al 2008a). While the overall result is a decline, the severity of decline depended on elevation. Low elevation sites had the largest declines and high elevations either had smaller declines or in some cases increased. Cuo et al (2009) analyzed 1915-2006 data from individual Puget Sound basins and confirmed the influence of elevation. They found relatively moderate snow water equivalent declines (up to 23%) in the Dosewallips, Nisqually, Puyallup, and Skagit basins, which are located at high elevations. However, other basins found in an intermediate elevation zone had declines greater than 30%.

Forecasting into the coming years by several studies show an agreement that continued loss of snowpack is to be expected as temperatures keep rising, although the amount of decline varies among studies. Estimated changes in Washington's snow water equivalent measurements associated with climate change depend on elevation with low elevations again expected to have the largest decrease. The Climate Impacts Group (2009) predicted the low-elevation snow water equivalent will decline in the range of 15-37% by 2020's, 23-54% by 2040's, and 39-71% by the 2080's. For the Washington Cascade Range only, Casola (2009) estimated a smaller decline in the range of 11-21% in snow water equivalent by 2050. This decline will likely affect water availability for both wildlife and people. For example, Puget Sound [Chapter2a.Salmonids#chinookanchor|Chinook salmon]] populations may have an increase of younger spawners and smaller proportions of stream-type fish (Beechie et al 2006) and the citizens of Puget Sound will incur declines in the municipal water supply and hydropower production (Climate Impacts Group 2009).

Along with the loss of snowpack, glaciers in the North Washington Cascade Range are also decreasing in volume and extent and predicted to continue to decrease. Pelto (2008) found significant changes in glacier mass balance, which is the difference between accumulation and ablation and advancing/retreating terminus behavior. The annual mass balance of ten glaciers was measured over two decades (1984-2006) and were found to have a 20-40% loss of their total volume. Furthermore, all 47 glaciers that were monitored are currently undergoing a significant retreat and four of them have disappeared. This trend of mass loss has accelerated in the last 15 years and is no longer dominated by shifts in the Pacific Decadal Oscillation, indicating recent large scale climate changes are stronger than the Pacific Decadal Oscillation induced variations of earlier decades of record (Josberger et al 2007). Loss in both snowpack and glaciers is expected to persist as global average temperatures continue to rise.

Range Shifts

The loss of snowpack and glaciers at mid and high-elevations constrains and also expands opportunities for animal and plant species settlement. In a warmer climate, species will begin to shift their ranges, or become less abundant in their current range in response to rising temperatures and precipitation shifts. Using Douglas-fir as an example, the Climate Impacts Group (2009) found that by the end of the 2060's climate will be sufficiently different from the late 20th century to alter Douglas-fir distribution in Washington State. Roughly 32% of the area currently classified as appropriate for Douglas-fir would be outside the identified climatic envelope for this species by the 2060s. This decline of suitable habitat mostly occurs at lower elevations due to water balance deficient. Currently, at high elevations Douglas-fir is constrained by snow and low temperatures (Griesbauer and Green 2010). With climate change predicted to cause warmer temperatures, less snowfall and earlier snowmelt, Douglas-fir may have increased productivity and expand its range to higher elevations (Griesbauer and Green 2010). Thus, it is unlikely that Douglas-fir in the Pacific Northwest will exhibit substantial range contractions unless water balance deficit increases substantially (Littell et al 2008).

More generally, Zolbrod and Peterson (1999) used a gap model to examine the effects of increased temperature (2°C) and altered precipitation on high-elevation ecosystems of the Olympic Mountains. They found in the southwest region, as tree species shift upwards in elevation with a warming climate, composition of tree species remains relatively stable. However, in the northeast region, the warmer climate results in combinations of tree species that is uncommon currently. Thus, this study suggests that species and site-specific responses at mesoscale and microscale resolutions must be characterized to quantify the variation in response of forest vegetation to climatic change.

Plants will not be the only communities to shift in response to a warming climate; wildlife too, will alter their range and abundance. Of 434 species worldwide that has been categorized as shifting in range, either measured directly at range boundaries or inferred from abundance changes within communities, 80% (P< 0.1 X 10-12) shifted in accord with climate change predictions (Parmesan & Yohe 2003).

One abundance/range shift of importance in the Salish Sea, particularly because it is an iconic group of species, is that of salmon. From the early 1800's to the late 1900's, the size of salmon runs declined by 92% in Puget Sound, 98.2% along the Washington Coast and 63.8% in British Columbia (Lackey 2003). Part of this decline may be attributable to rising stream temperatures, which cause a decrease in quality and quantity of salmon habitat.

Salmon are sensitive to thermal increases, with impairment occurring at the following stated temperature ranges for these different stages of their life history: smoltification and spawning 12-15°C, disease virulence 16°C, migration 19-23°C, and lethal threshold 24-26°C (Richter & Kolmes 2005). Simulations predict increasing freshwater temperatures and increasing thermal stress for salmon in western Washington that are slight for the 2020s but increasingly greater later in the 21st century (Climate Impacts Group 2009). Annual maximum temperature in the 2020s at most stream stations is projected to rise less than 1°C, but by the 2080s many stations warm by 2 to 5 °C (Climate Impacts Group 2009).

Not only do increased temperatures affect the health of salmon, they are also capable of impacting stock/population distribution. Stream temperatures at some sites in western Washington were high enough (21°C) for 10 weeks of the 1980's to prevent migration (Climate Impacts Group 2009). In the future, the persistence of water temperatures greater than 21°C is predicted to start earlier in the summer, and last later into the year than in previous decades (Climate Impacts Group 2009). Salmon thermal threshold level coupled with this temperature rise causes the projected loss of salmon habitat in Washington to range from 5 to 22% by 2090, depending on the climate change scenario used in the analysis (Climate Impacts Group 2009). The interaction of reduction of local riparian vegetation due to development with increased temperatures from a changing climate will likely exacerbate the loss of thermally appropriate salmon habitat, since riparian vegetation exerts a strong influence on buffering stream temperatures (Poole and Berman 2001).

Temperature increases also affect the abundance and distribution of less beneficial species in the region. Insect outbreaks can have substantial negative impacts on forest ecosystems by reducing growth and causing mortality (Kurz et al 2008). For these insects, warming is likely to cause elevation shifts and encourage northward expansion of the range of southern insects (Climate Impacts Group 2009; Parson et al 2001, Williams & Liebhold 2002). For example, low elevations will become unsuitable in a warming climate for Mountain Pine Beetle, and model simulations predict attacks will occur at increasingly higher elevations, lessening the amount of overall suitable habitat for outbreaks in western Washington (Climate Impacts Group 2009).

In another instance, the spruce budworm has been extending its range northward. Cool summer temperatures slow feeding and development of the larvae which increases its vulnerability to predators. Thus increased temperatures, coupled with drought stress (Parson et al 2001) diminish this limiting factor and allow for expanded populations. Using a simulated climate for years 2081-2100 predicts outbreaks being approximately 6 years longer with an average of 15% greater defoliation (Gray 2008).

Phenology

Not only will species ranges and distributions be affected but phenology, the seasonal timing of plant and animal life cycle events, is also affected by climate change. Worldwide, 677 species were quantitatively assessed in which 27% showed no trends in phenologies, 9% showed trends towards delayed spring events, and the majority, 62% showed trends towards spring advancement (Parmesan & Yohe 2003). Of these shift changes, an overwhelming majority of species examined (87%) occurred in the direction expected from climate change (P< 0.1 X 10-12). Another meta-analysis of 1,468 species found a comparable result with 81% (90% CI: 73.4–88.6%) of the shift changes occurring in the expected direction (Root et al 2003). Trends of early life cycle changes were observed in multiple taxa including; frog breeding, first flowering, tree budburst, bird nesting and arrival of migrant birds and butterflies.

Changes in phenology are important to ecosystem function because the level of response to climate change can vary across functional groups and several trophic levels. The decoupling of phenological relationships will have important implications by altering trophic interactions and causing eventual ecosystem-level changes. Studies performed here locally already show that decoupling is occurring. In Lake Washington, due to long-term climate warming and large-scale climatic patterns like Pacific decadal oscillation (PDO) and El Niño–southern oscillation (ENSO), phytoplankton spring bloom occurs 19 days earlier than it did in 1962, whereas the peak for zooplankton has advanced at either slower rates or remained stable (Winder & Schindler 2004).

These changes have created a growing time lag between the spring phytoplankton peak and zooplankton peak, which can be especially critical to *Daphnia*. In addition, Daphnia are a major zooplankton prey for the juvenile sockeye salmon in Lake Washington. Hampton et al (2006) found that the gap between the arrival date of juvenile sockeye and the spring peak onset of *Daphnia* has been increasing over the past nine years. Consequently, sockeye are forced to forage on less desirable and nutritious prey for longer periods of time. Such temperature driven phenological changes have the potential to severely impact the balance of native communities.

Placeholder: Additional phenological impacts, for example migration, wintering birds, pollinator/flowering timing.

Productivity

Rising temperatures in the future are predicted to increase overall forest productivity. However, this increase will not be uniform across elevations. Lower elevations will experience declines in productivity while an increase of productivity in many higher elevation forests partially offset those declines (Nakawatase & Peterson 2006; Latta et al. 2010). For example, Douglas-fir is limited by high growing season temperatures and low growing season precipitation at low to mid elevations (495–1133m), but at high elevation (1036-1450m) current-year high temperatures lead to above-average growth (Case and Peterson 2005).

Natural Hazards

Dry, warm conditions in the seasons leading up to and including the fire season are associated with increased area burned and more numerous fires in the western region of the United States (Heyerdahl et al 2008; Littell et al 2009). In the Puget Sound/Georgia basin region, even though there is an abundant fuel load, typically the climate has been a limiting factor for fires due to high moisture levels preventing ignition and spread (Bessie and Johnson 1995). However, with climate gradually becoming hotter and drier the frequency and intensity of fire is increasing. In the western U.S., wildfire frequency from 1987-2003 has increased roughly four times the average of 1970-1986 values, and the total area burned by these fires was more than six and a half times its previous level (Westerling et al 2006). This pattern is seen more specifically in the Western Cascade Range of Washington, with an average of 445 hectares (ha) burned from 1980-2006, with an expected increase to 769 ha by 2020's, 1295 ha by 2040's, and 3683 ha by 2080's based on statistical fire models that explain 50-65% of the variability in area burned (Climate Impacts Group 2009). Summer temperature, which the climate modeling community has high confidence in future predictions of, is the most important factor when considering the amount of area burned (Lawler and Mathias 2007). Thus, the projected increases in wildfire should be seen as highly likely and disturbance from fire will have an increased role in impacting forest communities and associated ecosystem services.

In the Salish Sea ecosystem, warmer climate, lower precipitation, reduced snowpack and earlier snowmelt along with increased vegetative activity, enhances soil drying and causes a decrease in summer soil moisture (Climate Impacts Group 2009). With the 50th percentile being equal to mean historical values, soil moisture is projected to decrease and be in the 38th to 43rd percentile by the 2020s, 35th to 40th percentile by the 2040s, and 32nd to 35th percentile by the 2080s. Although summers are predicted to be drier, the shift towards wetter autumns will have an impact on landslide frequency. Currently the highest landslide frequency along the southwest coast of British Columbia occurs during the autumn (Jakob & Lambert 2009). Models predict that on average, a 10% increase in 4 week antecedent rainfall and a 6% increase in 24-hour precipitation can be expected by the end of the next century (Jakob & Lambert 2009). This increased level of soil saturation during autumn suggests landslides will occur even more frequently than they do currently, but it is not clear what the magnitude of this increase will be.

Placeholder: Additional natural hazards, including storms

Pressure: Thermal Expansion

Globally, climate change is driving a thermal expansion of the world's oceans. When the air temperature rises, the ocean absorbs more of this heat. As the water temperature rises it also decreases in density which causes an expansion in volume; thus producing a rise in sea level. Since the circulation of the ocean slowly brings cold, deep water into contact with the increased thermal conditions at the surface, thermal expansion of the ocean will continue for roughly 1000 years after atmospheric temperature becomes stable (Mote et al 2008b).

Pressure: Melting of Land Ice

Global climate change is causing a decline of the world's glaciers and ice sheets (For details regarding Cascade Range glaciers see Impacts within Weather/Temperature Pressure sections above). The rate of change in land ice can be determined by looking at its mass balance. Mass balance is measured by determining the amount of snow accumulated during winter, and then measuring the amount of snow and ice removed by melting in the summer. The mass balance is the difference between these two measurements. Globally and locally the overall trend during the 20th century has been a decrease in the mass of land ice (IPCC 2007; Pelto 2008).

State: Sea Level

Sea level rise can result from either ocean thermal expansion, melting of land ice or both. Global sea level is rising due to these two factors, although each contributes a varying amount towards the overall rise. Antonov et al (2005) and Ishii et al (2006) both found similar rates of expansion of the world's oceans over the latter half of the 20th century. According to their research, the decades of 1955-2003 show sea level change of 0.33 ± 0.07 and 0.36 ± 0.06 mm yr–1 respectively. The last decade of this period, 1993-2003, shows sea level change of 1.2 ± 0.5 mm yr–1. However, more recent estimates of this same 1993-2003 period are slightly lower at 0.8 mm yr–1 (Domingues et al 2008). Meanwhile, glaciers and icecaps are estimated to have contributed to sea-level rise about 0.4 mm yr-1 from 1961 to 1990, increasing to 1.0 mm yr-1 from 2001 to 2004 (Church et al 2008). Thus, both thermal expansion and land ice melting seem to be increasing in rate going into the 21st century.

Projections into the 21st century by the IPCC fourth assessment report (2007) indicate that global sea level rise is expected to rise between 18 and 38 cm for their lowest emissions scenario, and between 26 and 59 cm for their highest emissions scenario. However, locally in the Puget Sound/Georgia Basin, sea level rise is determined by sea-level changes relative to the local land rather than the global average sea-level changes (Church et al 2008). The two global pressures (thermal expansion and melting of land ice) combine with local pressures (tectonic movement) to alter the state of the region's sea level, giving a relative sea level rise.

The rate and direction of tectonic movement varies across the Salish sea ecosystem (Climate Impacts Group 2009). The Northwest Olympic Peninsula has the highest rates of tectonic uplift, roughly 2 mm/yr. While the central and southern Washington coast have lower uplift rates of less than 1 mm/yr. South Puget Sound has seen an opposite trend and has been subsiding at a rate of 2 mm/yr.

Based on the rate and direction of tectonic shift as reported by the Climate Impacts Group (2009), coupled with the average of the six central values from the six IPCC scenarios, a medium advisory level of sea level rise by location in Washington State is given (Mote et al 2008b). By midcentury (2050) sea level is advised to increase by 0 cm in the Northwest Olympic Peninsula (range: -12 to +35 cm), 12.5 cm on the central/southern coast (range: +3 to +45 cm) and 15 cm in Puget Sound basin (range: +8 to +55 cm). Projecting out 50 years farther to 2100, sea level increases 4 (range: -24 to +88 cm), 29 (range: +6 to +108 cm), and 34 cm (range: +16 to +128) in the Northwest Olympic Peninsula, central/southern coast, and Puget Sound basin respectively.

However, it is stressed by Mote et al (2008b) that these calculations have not formally quantified the probabilities, sea level rise cannot be estimated accurately at specific locations, and these numbers are for advisory purposes and are not actual predictions.

Impacts: Although the magnitude of future sea level rise is uncertain, the major impacts are likely to be inundation, flooding, erosion and infrastructure damage. Sea-level rise leads to coastal flooding through direct inundation providing an increase in the base for storm surges, allowing flooding of larger areas and higher erosion rates. Sea level rise is predicted to increase erosion and flooding rates on the bluffs and beaches of the Puget Sound/Georgia Basin (Climate Impacts Group 2009), although the magnitude of change will depend on location and topography.

Sea level rise will cause the landward migration of the shoreline (and associated human enterprise and settlement) as waves break higher on the beach profile. While erosion is a natural episodic process, occurring mainly during infrequent events, such as storm surge waves during high tide, sea level rise will intensify this process. In general, for the region, beach erosion rates will vary depending on geomorphic characteristics, and extent of shoreline armoring (Finlayson, 2006). The Climate Impacts Group (2009) looked specifically at Bainbridge Island beaches. They found that locations most susceptible to inundation are the uplifted beach terraces on the southern third of the island and most of the islands bays and coves. About 48% of the shoreline is armored and NOAA recommends that unnecessary armoring structures, especially those that intrude into the intertidal zone, be either modified or removed. This is due to armoring generally causing a loss of sediment and shallow water habitat, which results in deeper water and higher energy waves which weaken the protective structure (See Increased Armoring in Shoreline Development section of this Chapter for further information).

Coastal bluffs will also be affected by sea level rise. Steep bluffs rim more than 60% of the Puget Sound shoreline, rising 15 to 150 meters (Johannessen & MacLennan 2007). Bluff erosion is a natural ongoing process that provides sediments to beaches. The erosion rate of a bluff is affected by geology, waves, and weather; thus varying amongst locations. Highest erosion rates, 2-10 cm/yr, are found in the Northern Straits because of greater wave exposure and poorly consolidated sediments. Common erosion rates farther south are on the order of a few centimeters a year, or less, in most locations.

The Climate Impacts Group (2009) looked specifically at the bluff erosion rates on Whidbey, Bainbridge and the San Juan Islands. Bluff erosion rates on Whidbey occur at a rate of one-61 cm/yr with landslides occurring frequently on the western shore. Bainbridge erosion rates vary between 5-15 cm/yr, with 20% of the shoreline classified as unstable. In contrast, the San Juan Islands with highly resistant bedrock bluffs, have relatively trivial erosion. These three sites illustrate the variety of responses expected to be seen as future sea level rises. Sites such as Whidbey and Bainbridge will be subject to increased hazards of erosion, landslides and damage, while sites like the San Juan Islands will be unlikely to be significantly affected due to the differences in substrate; sand for Whidbey and Bainbridge versus bedrock for San Juan Islands.

Placeholder: Infrastructure damage (ex: stormwater and wastewater)

Placeholder: Impacts of sea level rise on distribution of human population

Pressure: Large and local scale atmospheric forcing

In the Pacific Northwest, El Niño/Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) are both large scale patterns of hemispherical climate variability involving sea surface temperature fields that each create comparable warm and cool phases (Moore et al 2008a). However, the PDO phases persist for 20-30 years, whereas the ENSO only occurs for 6-18 months (Mantua and Hare 2002). The translation of ENSO and PDO related changes into observable changes in oceanographic properties can be variable and indicates local forcings are also involved. Thus, one local climate forcing parameter, surface air temperature, is found to be the primary cause of variability in the temperature of the Puget Sound, with effects of ENSO and PDO being secondary (Moore et al 2008a). In particular, winter is the season with the greatest coupling of both local and large scale forcings, with sea surface temperature having significant correlations to all scales of forcings during this season (Moore et al 2008a).

Placeholder: More detail on circulation, local weather and winds as forcings of SST.

State: Sea Surface Temperature

Globally, observations since 1961 show that while land regions have warmed faster than the oceans, the ocean has been taking up over 80% of the heat being added to the climate system and the average temperature of the global ocean has increased to depths of at least 3000m (Field et al 2007). This pattern holds true for the Pacific Northwest, where modeled sea surface temperature is 1.2°C higher, which is less than land area warming (2.0°C), but is still a significant increase relative to the inter-annual variability of the ocean (Climate Impacts Group 2009).

Impacts:

The coastal sea surface temperature of the Pacific Northwest helps determine the biological and physical conditions of the marine environment and estuaries. The Climate Impacts Group (2009) expects that by the year 2100, surface water temperatures in Puget Sound will increase by roughly 6°C. Since temperatures higher than 13°C promote harmful algal blooms, an increase of this magnitude is likely to cause earlier and longer lasting blooms. For example, from 1921-2007, the planktonic dinoflagellate *Alexandrium catenella*, which is associated with paralytic shellfish poisoning, had a 68 day window where temperatures reached the 13°C threshold for optimal growth (Moore et al 2008b). Scenarios for warmer sea surface temperature conditions in the future of 2, 4, and 6°C will expand that optimal window by 69, 127, and 191 days respectively.

Placeholder- productivity, higher trophic level impacts, phenological impacts (migration, spawning), hypoxia

Pressure: Ocean Acidification

Atmospheric CO_2 concentration is approximately 387 parts per million by volume (Le Quéré et al 2009). This level has not been reached in at least 650,000 years, and it is projected to increase by 0.5% per year (Guinotte and Fabry 2008). In recent decades, only half of anthropogenic CO_2 has remained in the atmosphere; of the remaining half, 20% has been taken up by the terrestrial biosphere and 30% by the oceans (Feely et al 2004). As the global ocean absorbs atmospheric carbon dioxide, these increasing concentrations are reducing ocean pH and carbonate ion concentrations, resulting in the oceans' acidification (Orr et al 2005).

State: Increased Ocean pH

Since the Industrial Revolution, the global ocean surface pH has dropped by 0.1 pH units (Guinotte and Fabry 2008). This corresponds to approximately a 30% increase in hydrogen ion concentration. According to Feely et al. (2004) by the end of the century, estimates of atmospheric and oceanic CO2 concentrations are predicted to be over 800 ppm. Additionally the level of ocean surface dissolved inorganic carbon would increase by 12%, with carbonate ion concentration decreasing by about 60%. The associated drop in pH would be roughly 0.4 units in surface waters.

In Puget Sound, acidification accounts for 24% of the pH decrease in the summer and 49% in the winter relative to preindustrial values (Feely et al 2010). Under the predicted doubling of atmospheric CO2 levels by the end of the century, the contribution of acidification on the decrease in pH would increase to 49% in the summer and 82% in the winter (Feely et al 2010).

Impacts:

Depth offers no protection from ocean acidification; the deepest communities, such as cold-water corals in each ocean will be the first to experience a shift from saturated to unsaturated conditions and will contract in vertical depth distribution (Doney et al 2009). By 2100, 70% of cold-water corals, key refuges and feeding grounds for commercial fish and shellfish species, will be exposed to acidified waters.

Along the west coast of Washington, the seasonal upwelling of acidified deep water reaches depths of 40-120m on the continental shelf (Feely et al 2008). While this is a natural phenomenon in the region, the oceanic uptake of anthropogenic CO2 has increased the areal extent and the potential threat of these acidified waters to many calcifying species that live along the coast. Increasing ocean acidification reduces the availability of carbonate minerals, important building blocks for marine plants and animals, and thus reduces the rate of calcification (Andersson et al 2008). Data from multiple studies compiled by Fabry et al (2008) indicate that foraminifera, molluscs, and echinoderms demonstrate reduced calcification and sometimes dissolution of CaCO3 skeletal structures when exposed to decreasing pH conditions. Ocean acidification may cause these calcareous marine species to decline, and be replaced by non-calcareous counterparts (Wootton et al 2008) altering the food web and community structure.

Placeholder- Detailed information on shifts in species dominance and community composition, altered food webs. ''' Pressure: UV Irradiance'''

UV radiation is classified as UV-A (315–400 nm), UV-B (280–315 nm), and UV–C (100–280 nm) (Kerr and Fioletov 2008). The shorter the wavelength, the more harmful it becomes to species health. If adequate amounts of ozone are present in the atmosphere, it effectively cuts off shortwave radiation at 290nm. Thus, there are important effects of changes in the intensity of solar UV-radiation resulting from stratospheric ozone depletion, particularly UV-B radiation (Solomon 2008). Since ozone strongly absorbs the radiation at UV wavelengths detrimental to most biological species, a decrease in stratospheric ozone could have a significant impact on the biosphere (Kerr and Fioletov 2008).

State: Increased UV Radiation

There are variations in incident UV radiation as a function of latitude and longitude, as well as major inter-hemispheric differences for the same latitude and season over the ocean (Ahmad et al. 2003) It is estimated that for every 1% decrease in stratospheric ozone, there is a 1% to 2% increase in UV-B transmitted to the ocean (Zhou et al 2009). In the Pacific Northwest, for UV wavelengths of 380nm and 310nm, the maximum depth limit for UV biological effectiveness based on the absorptive properties of pure ocean water plus the added absorption and scattering of dissolved and suspended materials is 30 to 40 meters (Ahmad et al. 2003).

Impacts:

Placeholder- reduced productivity by phytoplankton and submerged vegetation

Data Gaps and Uncertainties

A major uncertainty associated with future climate change predictions are the future emission levels of greenhouse gasses. This uncertainty can be partially alleviated by assessing multiple scenarios of various intensities of radiative forcing, for example the Climate Impacts Group (2009) used 20 such models in their predictions. However, uncertainty in how the climate system will respond is still prevalent. Zickfeld et al (2010) asked 14 leading climate scientists what contributes most to the uncertainty associated with different radiative forcing scenarios. The scientists ranked cloud radiative feedback as the factor contributing the most to uncertainties in future global mean temperature change for all scenarios. In addition, the climate scientists expect that even with new research there will only be modest reductions in uncertainty over the next 20 years (Zickfeld et al 2010). These uncertainties should be considered with developing management responses.

Driver: Residential, Commercial and Industrial Development

Perhaps the single greatest source of transformation in the Salish Sea terrestrial ecosystem is the conversion of lowland forests to a mosaic of residential, commercial and industrial lands created for human use. The state population, currently at over 6 million people, doubled over the last 40 years and is predicted to reach 8 million by 2030 (WOFM 2010). The highest population density is within the Puget Trough region (WOFM 2010). Changes in the landscape are driven by expanding human populations associated with growing businesses (e.g., Microsoft, Amazon.com, Boeing) and rich natural amenities (Alberti 2008), and changing family structure (single parent households associated with high divorce rates have greatly increased the demand for residential dwellings; Morrill 1992—see http://faculty.washington.edu/morrill/). Populations are expanding in the cities and exurban environments (Alberti et al. 2003; Alig et al. 2004; UNFPA 2007; WOFM 2007, 2010; Alberti 2008; Grimm et al. 2008). Increasing population growth results in more roads, more industry, more houses, more transportation, and more businesses. These changes mean positive changes to income, business growth, etc. However, we only focus in the current draft on the terrestrial ecological changes resulting from residential, commercial and industrial development rather than the benefits to human health and well-being; additionally, negative impacts of ecological changes on human health and well-being, such as impaired water quality and decreased resource availability, are currently omitted. Effects of development on nearshore ecosystems are discussed separately (see “Threat: Shoreline Modification”). We also treat agriculture separately due to the distinct set of pressures, states and impacts associated with this distinct type of development (see “Threat: Agricultural Practices (Placeholder)”).

Development-related land uses associated with residential, employment and commercial activities span a gradient of density and intensity from compact, highly developed urban, commercial and industrial centers to the more sparsely developed exurban and rural fringes (Alberti et al. 2004; Pickett et al. 2008). These drivers result in a diverse range of ecological effects (Alberti 2005). Development broadly encompasses low- to high-density housing as well as retail stores and businesses, industrial production and storage facilities, and transportation infrastructure. In the Driver-Pressure-State-Impacts-Response (DPSIR) model, these activities are represented in the “Drivers” (Figure 2).

Below, we review the Pressures associated with residential, commercial and industrial development, and the resultant State changes and system Impacts. Our primary focus in the current draft is on States and Impacts resulting from land use/land cover change as a Pressure in the Salish Sea ecosystem. Such effects manifest themselves at multiple levels, from ecosystems to species, and within both terrestrial and aquatic systems. States and Impacts associated with the Pressure of infrastructural demands will be addressed in future revisions. Increased chemical inputs, both naturally and anthropogenically derived, are a significant development associated Pressure (Figure 2). Chemical contaminants are reviewed more broadly and fully under “Threat: Pollution.” Additionally, Chapter 4, Effectiveness of Strategies to Protect and Restore the System, addresses the human Response to the problems associated with development, and will not be covered in this chapter.

Placeholder – positive and negative impacts of residential, commercial and industrial development on human health, socioeconomics and overall well-being

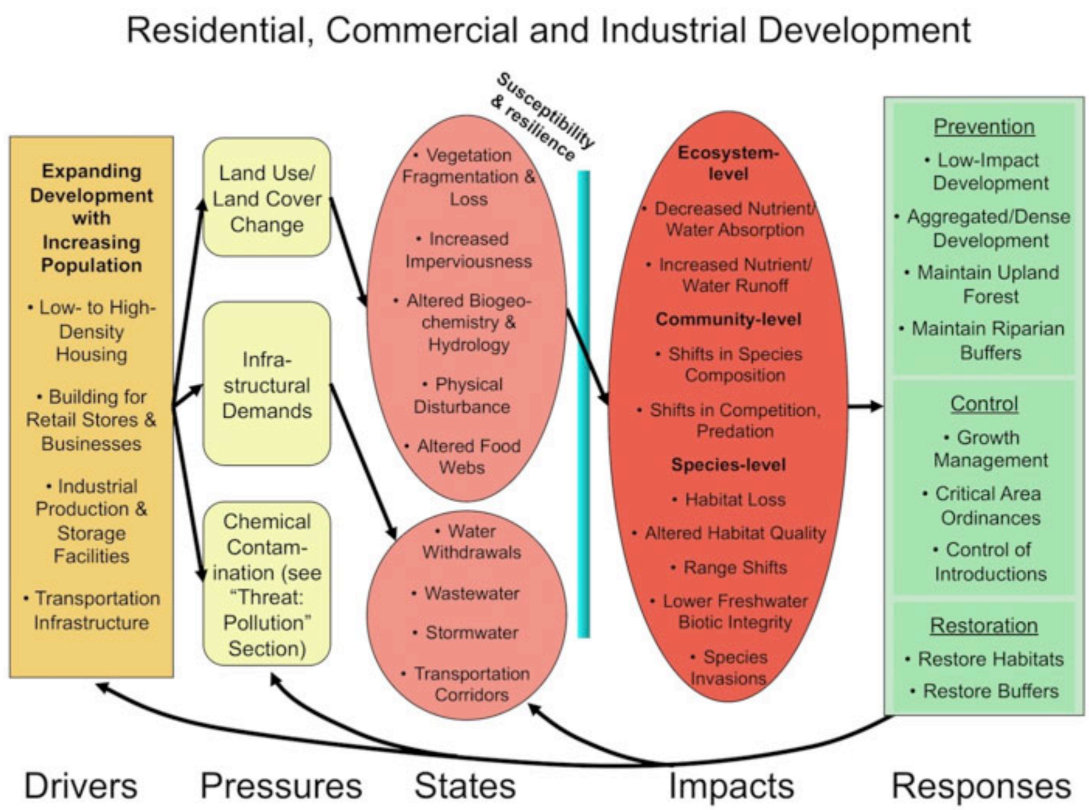

Figure 2. Driver-Pressure-State-Impacts-Response conceptual model for residential, commercial and industrial development in the Salish Sea ecosystem.

1. Pressure: Land Use/Land Cover Changes

Development is most visibly characterized by significant land use and land cover (LULC) transformations. In terrestrial systems of the Salish Sea region, forests, wetlands, prairies and agricultural lands are converted to residential, commercial and industrial uses. The rates of conversion have increased significantly in the late 20th century and are projected to continue to increase (Alig et al. 2004; Alberti 2005; Robinson et al. 2005; White et al. 2009). The region most heavily impacted by the human footprint in Puget Sound is in the central Sound: the amount of developed lands increased from 16% to 23% of the total area between 2002 and 2007 (Alberti et al. 2004; Hepinstall et al. 2008), an increase of approximately 1.4 percent per year. Continued development at this rate will result in developed lands extending well into the Cascade Mountain foothills by 2027 (Hepinstall et al. 2008).

Transformations of both land cover composition and configuration in the Puget Sound watershed, particularly in the central Sound region of Snohomish, King, Pierce and Kitsap Counties, have been extensive. In an analysis of LULC change in central Puget Sound, Alberti et al. (2004)

measured a 6.7 percent increase in paved urban areas and a 7.8 percent increase in mixed urban areas from 1991 to 1999. Largely associated with increasing urbanization in the region, almost half of the land converted to urban land uses occurred in the Seattle metropolitan region, and included significant conversion of adjacent forests (Alberti et al. 2004). Similarly, Hepinstall et al. (2008) examined trends in LULC change in the central Puget Sound region from 1991-2002, and developed a model to forecast future trends of change. Between 1991-1995, observed annual rates of agriculture and coniferous forest loss were 8.0 and 2.3 percent per year, respectively. From 1995-1999, rates of agriculture loss slowed to 1.3 percent per year and coniferous forest began to show an increase of 1.0 percent per year (mostly as a result of regenerating forests in the uplands), but mixed deciduous-coniferous forest declined by 4.7 percent per year. By spatially extrapolating these trends into the future, Hepinstall et al. (2008) forecasted mature forest cover composition will decline from approximately 45% in 1999 to 27-30 percent of the total central Sound area by 2027. Significant native vegetation cover still remains in the Puget Basin: 53 percent of the central Puget Sound region was still composed of forested lands (down from 66 percent in 2002; Alberti 2009). However, the above trends suggest that land conversion, particularly of forests, has occurred and continues to occur at a considerable rate.

State: Vegetation Fragmentation and Loss

The most dramatic examples of LULC change result in the loss and fragmentation of plant cover. Fragmentation and loss describe two interrelated facets of landscape composition and pattern (Turner et al. 2001; Alberti and Marzluff 2004). Loss of native vegetation results from replacement by other land cover types, particularly those associated with residential development (e.g., buildings, roads, and planted landscapes). This loss affects land cover composition and changes ecosystem processes (Fahrig 1997; Alberti and Marzluff 2004; Wiegand et al. 2005; Donnelly and Marzluff 2006). Additionally, fragmentation can introduce sharp ecotones or edges that affect both material flows in ecosystems (Wickham et al. 2002; Walsh et al. 2005) and habitat conditions for species (deMaynadier and Hunter 2000; Hansen et al. 2005), particularly when there is a strong contrast between adjacent land cover types (e.g., impervious surface adjacent to forest). Since the vegetation loss and fragmentation are generally correlated and their interactions are difficult to untangle, we discuss their combined effects.

Impacts:

Development-related LULC change leads to impacts across and between scales, from the landscape and ecosystem level down to the species level. Most readily apparent are changes in the spatial pattern and configuration of landscapes such as the fragmentation of forests. Landscape fragmentation impacts ecosystems at multiple scales and levels of organization: it affects the distribution and persistence of species (Wiegand et al. 2005; Donnelly and Marzluff 2006), as well as fluxes of nutrients and water (Baker et al. 2001b; Wickham et al. 2002; Walsh et al. 2005). Even regions of low-density development, in which a significant percent of the landscape is comprised of forest, can exhibit significant levels of fragmentation due to inclusions of roads, houses, and other structures (Hansen et al. 2005). Bisection of (forest) habitats by roads has significant population-level impacts on many species (deMaynadier and Hunter 2000; Steen and Gibbs 2004), particularly for those who are attracted to habitats near or that regularly cross roads and are struck by vehicles (Fahrig and Rytwinski 2009). Human-induced and -maintained land cover characteristics such as lawns and power transmission corridors modify biophysical structure and biogeochemical fluxes (e.g., Kaye et al. 2006) and negatively affect the persistence of native species assemblages (e.g., Alberti and Marzluff 2004; Hansen and Clevenger 2005). The specific spatial characteristics of fragmentation and its associated impacts are generally dependent on the intensity of development, which ranges from urban centers to rural fringes, (Alberti 2005; Alberti et al. 2007). Therefore, the threats to ecosystems not only result from the amount of vegetation conversion but also the resulting spatial pattern of the vegetation.

Entire ecosystems and ecological communities are threatened by LULC changes and associated impacts. For example, western Washington's native grasslands and oak woodlands have declined to less than 3% of the pre-European settlement areal extent (Crawford and Hall 1997). Factors contributing to their decline and degradation include urban and agricultural conversion, fire suppression, conifer tree invasion and invasion by non-native and invasive species (Giles 1970; Agee 1993; Clampitt 1993; Crawford and Hall 1997; Chappell et al. 2001). The prairies and oak woodlands of western Washington are composed of eight international vegetation classification plant associations that are now critically globally imperiled or globally imperiled (Washington Department of Natural Resouces 2007; Natureserve 2008). As a result, many species of plants and animals associated with these ecosystems are also of conservation concern because of population declines, local extirpation, or close associations with declining plant communities including the golden paintbrush (*Castilleja levisecta*), Taylor's checkerspot butterfly (*Euphydryas editha taylori*), streaked horned lark (*Eremophila alpestris strigata*), and mazama pocket gopher (*Thomomys mazama*) (Dunn and Ewing 1997; Stinson 2005; Camfield et al. 2010).

Placeholder – ecosystems that are most threatened or have been lost

The loss of extensive, contiguous mature forest ecosystems is one of the most significant consequences of LULC change associated with development. Changes in the composition and configuration of landscapes result in significant changes to biogeochemical and hydrologic cycling. Vegetation and soils within forested ecosystems mediate the cycling of nutrients and water. As vegetation composition and pattern changes with increasing development, these ecosystem functions are altered. However, because the changes in biogeochemistry and hydrology that result from development go beyond the impacts of vegetation fragmentation and loss, we discuss the specific impacts in greater detail below (see "State: Altered Biogeochemistry and Hydrology).

A number of studies demonstrate the impacts of vegetation fragmentation and loss on instream biotic conditions, highlighting the existing linkages between terrestrial and freshwater ecosystems. Expanding on a previous study of urban land use impacts on biotic integrity (Booth et al. 2004), Alberti et al. (2007) examined relationships between landscape composition (directly related to vegetation amount) and configuration (including levels of fragmentation and edge contrasts) in the central Puget Sound and benthic indices of biotic integrity. They found a strong negative relationships between benthic indices of biotic integrity and contiguity of urban land cover, a somewhat weaker negative relationship with overall imperviousness, and still weaker but significant negative relationships with road density and road crossings. They observed these relationships between benthic indices of biotic integrity and landscape pattern both at the level of drainage basins and within 100-300 m buffers around streams. Refined and expanded observations by Shandas and Alberti (2009), however, determined that within the immediate vicinity of streams, instream biota are affected by the percent vegetation cover, not the configuration of vegetation. Collectively, these measures of stream biotic integrity (Morley and Karr 2002) relate significantly to overall water quality conditions (see below for further discussion of such impacts) as a function of development-related landscape changes. Implications of these studies point to the potential effectiveness of increasing the amount of upland vegetation and connectivity for mitigating downstream flow rates and volumes, particularly as result of high imperviousness in urbanized landscapes.

Placeholder – impacts of vegetation fragmentation and loss on riparian and stream ecosystems

Some important/useful references and information to include in this subsection:

- Key publications out of the River History Project and the various ages of the Water Center
  - Beechie, T., B. D. Collins, and G. Pess. 2001. Holocene and recent changes to fish habitats in two Puget Sound basins. In: J. M. Dorava, B. Palcsak, F. Fitzpatrick, and D. R. Montgomery, eds. Geomorphic Processes and Riverine Habitat. American Geophysical Union, Washington, D. C. pp. 37-54.
  - Collins, B. D., and D. R. Montgomery. 2002. Forest development, wood jams and restoration of floodplain rivers in the Puget Lowland. Restoration Ecology 10: 237-247.
  - Collins, B. D., D. R. Montgomery, and A. D. Haas. 2002. Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries & Aquatic Sciences 59: 66-76.
  - Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. In: S. V. Gregory, K. L. Boyer, and A. M. Gurnell, eds., The Ecology and Management of Wood in World Rivers, American Fisheries Society Symposium 37. American Fisheries Society, Bethesda, MD. pp. 21-48.
- fundamental shifts in vegetation from conifer dominated to alder and other deciduous and herbaceous vegetation along the shores of Lake Washington (Davis, M. B. 1973. Pollen evidence of changing land use around the shores of Lake Washington. Northwest Science 47:133–148)
- effects of alder on instream nutrient levels (Volk, C. J., P. M. Kiffney, R. L. Edmonds. 2003. Role of riparian red alder (Alnus rubra) in the nutrient dynamics of coastal streams of the Olympic Peninsula, WA, U.S.A. American Fisheries Society Special Publication 34: 213-228.)
- effects of urbanization and changing riparian vegetation on nutrient inputs to small streams (Roberts, M.L. and R.E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. JNABS 28:941-954.)
- Two volumes of JNABS focused on urbanization (v 24 and v 28):
  - Booth, D. B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of the North American Benthological Society 24:724-737.

- Brown, L. R., T. F. Cuffney, J. F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A. H. Bell, and J. T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051-1069.
- Carter, T., C. R. Jackson, A. Rosemond, C. Pringle, D. Radcliffe, W. Tollner, J. Maerz, D. Leigh, and A. Trice. 2009. Beyond the urban gradient: barriers and opportunities for timely studies of urbanization effects on aquatic ecosystems. Journal of the North American Benthological Society 28:1038-1050.
- Feminella, J. W. and C. J. Walsh. 2005. Urbanization and stream ecology: an introduction to the series. Journal of the North American Benthological Society 24:585-587.
- Grimm, N. B., R. W. Sheibley, C. L. Crenshaw, C. N. Dahm, W. J. Roach, and L. H. Zeglin. 2005. N retention and transformation in urban streams. Journal of the North American Benthological Society 24:626-642.
- Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24:602-612.
- Morgan, R. P. and S. E. Cushman. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society 24:643-655.
- Roberts, M. L. and R. E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. Journal of the North American Benthological Society 28:941-954.
- Roy, A. H., M. C. Freeman, B. J. Freeman, S. J. Wenger, W. E. Ensign, and J. L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society 24:656-678.
- Roy, A. H., A. H. Purcell, C. J. Walsh, and S. J. Wenger. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society 28:908-910.
- Smith, R. F., L. C. Alexander, and W. O. Lamp. 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. Journal of the North American Benthological Society 28:1022-1037.
- Steuer, J. J., J. D. Bales, and E. M. P. Giddings. 2009. Relationship of stream ecological conditions to simulated hydraulic metrics across a gradient of basin urbanization. Journal of the North American Benthological Society 28:955-976.
- Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society 24:690-705.
- Walsh, C. J. and J. Kunapo. 2009. The importance of upland flow paths in determining urban effects on stream ecosystems. Journal of the North American Benthological Society 28:977-990.
- Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Marti, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramirez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban

stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080-1098.

Placeholder – impacts of vegetation fragmentation and loss on downstream estuarine/marine ecosystems

Vegetation fragmentation and loss also impact the biodiversity and species composition of the region. A series of studies (Donnelly and Marzluff 2004, 2006; Blewett and Marzluff 2005; Marzluff 2005; Hepinstall et al. 2008, 2009) examined avian species richness and abundance along the urban-to-rural gradient in the Seattle metropolitan region. Overall diversity was highest at 40-60 percent forest cover, with the abundance and richness of native forest bird species decreasing with decreasing forest cover, and with synanthropic species (i.e., those that thrive in human-dominated environments) and, to a lesser degree, early successional species increasing at higher levels of development. Intermediate levels of forest cover (and greatest levels of fragmentation), characteristic of low density residential development and rural fringes, provide sufficient habitat for native forest species along with edge habitats and resource supplements favorable to early successional and synanthropic species (Donnelly and Marzluff 2004, 2006; Blewett and Marzluff 2005; Hansen et al. 2005; Marzluff 2005; Withey and Marzluff 2009). Overall declines in biodiversity occur at high levels of urbanization and forest loss (Donnelly and Marzluff 2004, 2006; Hepinstall et al. 2008, 2009; Whittaker and Marzluff 2009). It is important to note in this system, as in many ecological systems, diversity is increased by fragmentation of uniform land covers so that many distinct types of habitats are found in close proximity. When either forest or intensively built urban land dominates an area, diversity decreases. In addition, as the distance to neighboring forest reserves increases and/or the extent of such reserves decreases with increasing development, urban and suburban bird populations are likely to decline dramatically (Marzluff et al. 2007). Projecting such trends into the future, Hepinstall et al. (2008, 2009) forecast reduced species diversity with the spread of development in the Puget trough, with sharper declines in forest and early successional species when forest cover is reduced below approximately 40 percent, and as developed areas become older and more established (hence losing their successional characteristics).

Enhanced food and habitat choices for early successional and synanthropic species, associated with lower development densities, result in community level changes. Withey and Marzluff (2009) examined the relationships between American crow (Corvus brachyrhynchos) abundance and activity levels and land cover composition and configuration at three spatial scales in King County. Crow abundance at site (up to 200 ha) and within-site (approximately 18 ha) scales was strongly associated with mixed LULC characteristics: developed lands provide access to plentiful anthropogenic food resources while adjacent urban forest/maintained vegetation patches that provide access to insects and songbird nestlings and suitable nesting sites. At more localized scales of 400 m2, crows use the range of cover types relatively evenly. While, increased heterogeneity and edge habitats often result in increased nest predation by corvids, raptors, squirrels and raccoons, such effects have not been documented in the Salish Sea ecosystem (Marzluff et al. 2007). In fact, reduced predation in some urban settings has been shown to positively impact populations of urban birds which, in turn, resulted in top-down trophic effects on insect herbivory (Faeth et al. 2005).

Analogous shifts in predator-prey dynamics and trophic relationships occur with urban coyote (*Canis latrans*) populations. Landscape heterogeneity combined with supplemental anthropogenic food resources – including house cats – in urban ecosystems provide favorable habitat conditions for coyotes in the Puget Sound region (Quinn 1997a,b) and other urban settings (Crooks and Soulé 1999; Crooks 2002; Patten and Bolger 2003; Gehrt and Prange 2007). Observations do not yield uniform conclusions regarding these trophic interactions (Gehrt and Prange 2007) and illustrate the important role of specific species-habitat relationships (Patten and Bolger 2003) in determining such interactions. Nonetheless, coyotes feed on mesopredators such as cats and raccoons (Quinn 1997a), which can have indirect positive impacts on urban songbird productivity (Crooks and Soulé 1999; Crooks 2002).

Placeholder – impacts on amphibian species

Placeholder – impacts on fish species

Placeholder – impacts on marine mammals

Placeholder – impacts on breeding versus non-breeding and resident versus migratory populations

State: Increased Imperviousness

Changes in LULC associated with residential, commercial and industrial development result in changes to hydrologic and material fluxes, volumes and pathways. At the more extreme level is the replacement of native vegetation with impervious surfaces: roadways, building structures, and artificial drainage pathways. Levels of imperviousness in urban landscapes result in modified surface- and groundwater pathways, water filtration and flow rates, which disrupts the balance between ground and surface water flows. Consequently, flows are linearized, more directly transported into streams and water bodies and result in more abrupt, extreme peaks in stream flow rates and volumes after storm events (Tague and Band 2001; Booth et al. 2002; Kaye et al. 2006). These modified pathways take both direct forms (e.g., culverts and stormwater drainage systems), and indirect forms (e.g., roads, building roofs, parking lots, and other built structures) that divert and focus water flow.

One of the significant characteristics of impervious surfaces is their relative permanence. Once constructed, residential, commercial and industrial structures tend to remain in place or are replaced by new impervious structures (Alberti et al. 2004; Alberti 2008). Alberti et al. (2004) noted that 86 percent of the central Puget Sound region consisting of paved land cover in 1991 was in the same state 8 years later. For mixed urban classes, which comprise between 15-75 percent impervious surfaces, persistence from 1991 to 1999 was approximately 96 percent (Alberti et al. 2004).

Impacts:

As a result of increasing imperviousness associated with development, water, nutrients, bacteria, toxics and pollutants that would be absorbed, filtered and channeled by soils and vegetation in forested watersheds are more likely to be transported directly, more acutely, and in larger volumes, to streams, rivers, lakes, and ultimately the Salish Sea. By definition, impervious surface cover also impairs or prohibits the infiltration of water and nutrients into soils by creating an impermeable barrier over soils and by compacting remnant soil layers (Ragab et al. 2003a,b; Gregory et al. 2006; Kaye et al. 2006). Managed lawns also act as semi-pervious, if not impervious, surfaces, despite their vegetative nature: they have shallow, densely packed rootmats that result in compacted soils that reduce permeability relative to native forest communities (Schueler 1995; May et al. 1997).

Placeholder – expanded discussion of impervious surface impacts on hydrology and soils needed; e.g., C. P. Konrad, D. B. Booth, and S. J. Burges, 2005, Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance: Water Resources Research, v. 41(7), W07009, doi:10.1029/2005WR004097. See also other Water Center studies.

Placeholder – expanded discussion of altered soil conditions such as compaction and reduced absorption

Increased runoff from impermeable surfaces results in rapid and significant discharge of water, often highly contaminated into the Salish Sea. In the more extensively developed watersheds of central Puget Sound, stream gauge data indicate extensive fluctuations around annual mean daily flow volumes, and frequent occurrences of volumes above such levels . Krahn et al. (2007) attribute levels of persistent organic pollutants (POPs) occurring in resident Puget Sound marine mammals to direct transport of contaminants to water bodies, a consequence of imperviousness (Booth et al. 2002). A significant proportion of waterbodies in Washington listed as impaired for one or more pollutants under Section 303(d) of the Clean Water Act fall within the most developed regions that also have the highest impervious cover in the Puget Sound lowlands (Alberti et al. 2004).

Increased water and contaminant runoff from impervious surfaces have significant implications for biotic conditions in the Sound/Basin ecosystem (see Pollution threat below). For example, Bilby and Mollot (2008) observed significant relationships between changes in land cover and coho salmon (*Oncorhynchus kisutch*) abundance. Urbanizing watersheds in the central and northern Sound, which in the late 1980's provided habitat for approximately 20 percent of the total number of spawning fish, were occupied by less than 5 percent of total fish numbers by around 2000. They attribute these shifts to heightened runoff resulting from increased imperviousness, leading to both higher water flow rates and volumes and increased mobilization of contaminants relative to levels observed in watersheds dominated by rural residential and forested areas (Bilby and Mollot 2008). Benthic indices of biotic integrity (Morley and Karr 2002; Booth et al. 2004; Alberti et al. 2007; Shandas and Alberti 2009) and fish (Matzen and Berge 2008) have declined as a consequence of the percent imperviousness within watersheds.

Placeholder – impacts on other fish species, zooplankton, and broader food web structure and function

State: Altered Biogeochemistry and Hydrology

Changes in the vegetation structure within watersheds alter or remove the water and nutrient retentive capacity associated with intact forests (Tague and Band 2001; Wickham et al. 2002; Groffman et al. 2004, 2006; Kaye et al. 2006). Ecological functions performed by remnant urban forest patches are diminished relative to their more connected, structurally and biologically complex exurban counterparts. Urban forests, for instance, exhibit higher potentials for nitrogen saturation (Wickham et al. 2002; Groffman et al. 2004; Zhu and Carreiro 2004). Water absorption into soils is also diminished or locally eliminated, particularly with higher levels of imperviousness. Changes in geomorphology and biota within urban riparian soils have been shown to lower denitrification potentials and thereby increase fluxes of nitrates into streams (Groffman et al. 2002, 2003, 2005). With losses of forest vegetation due to urban development, carbon sequestration will decline, with significant broader-scale implications for climate change (Churkina et al. 2010; Hutyra et al., in press). It should be noted, however, that recent research on forests along an urban-to-rural gradient in Seattle has pointed to significant carbon storage capacity within even urbanizing landscapes (Hutyra et al., in press).

LULC changes result in additional sources and inputs of nutrients. Fertilization of residential and recreational lawns contributes to increased soil nitrogen concentrations and runoff levels (King and Balogh 2001; Valiela and Bowen 2002; Law et al. 2004; Hope et al. 2005; Toran and Grandstaff 2007). Pet waste has also been suggested to be a significant component in urban nitrogen budgets (Baker et al. 2001a). Atmospheric deposition of nitrogen is typically higher in urban areas as a result of transportation- and industry-related combustion activities (Vitousek et al. 1997; Valiela and Bowen 2002; Kaye et al. 2006).

Impacts:

Collectively, the above changes in material fluxes and concentrations contribute to increased pollution and sedimentation in streams. Brett et al. (2005) examined LULC-dependent contributions of nutrients and sediments to stream concentration levels along an urban-to-rural gradient in the central Puget Sound. They examined relationships between biophysical characteristics, such as land cover, topography and soils, and nutrient and sediment concentrations within 17 subbasins of the Cedar/Sammamish Water Resource Inventory Area. Compared with more completely forested basins, urban streams exhibited roughly 40% higher nitrogen levels and approximately 110% higher phosphorus levels. They note that though these nutrient discharge levels are lower than what might be observed within agricultural regions (e.g., Wickham et al. 2002; Weller et al. 2003), the levels have significant non-point source pollution implications.

Development-related LULC changes also alter hydrologic flow rates and volumes, particularly through the introduction of impervious surfaces (see above). Booth et al. (2002) examined impacts of development-related modifications to hydrology in King County, WA, particularly in the context of stormwater runoff. They found that, as a consequence of altered hydrologic conditions, hydrographs for urban streams exhibited peak discharge rates that are as much as twice as high as under pre-development conditions. Beyond the immediate, direct impacts of imperviousness on urban hydrology, Booth et al. (2002) noted that upstream rural development can also have a significant impact on downstream water quality and quantity and stream channel stability, through land clearing and removal of riparian vegetation. Their results emphasize the importance of limiting imperviousness within hydrologically sensitive segments of drainage basins, but also the relatively more significant contribution that can be made by maintaining significant upland forest cover (e.g., through clustered development) and riparian vegetation (see also Baker et al. 2001b).

Cuo et al. (2009) compared hydrologic effects associated with lowland urban development to upland forest harvesting using a version of the Distributed Hydrology-Soil-Vegetation Model (DHSVM; see also Cuo et al. 2008) along with historic and current land cover and meteorological data. The hydrology of upland basins subject to forest harvest remained largely intact but with decreased evapotranspiration and faster snowmelt rates. Earlier snowmelt trends in the early 21st century, a function of shifting temperatures, have also led to decreased summer flows in upland regions. In lowland watersheds LULC change increased flows due to changes in infiltration and surface flows associated with urban development. Relative increases in flow rates and volumes in the lowland sites depended on the level of development within specific basins.

Placeholder – expand discussion of impacts on riparian and stream ecosystems; additional references and information for effects of urbanization on stream hydrology and geomorphology

Useful references:

- D. B. Booth, 2005, Challenges and prospects for restoring urban streams: Journal of the North American Benthological Society, v. 24, pp. 724-737.
- C. P. Konrad, D. B. Booth, and S. J. Burges, 2005, Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance: Water Resources Research, v. 41(7), W07009, doi:10.1029/2005WR004097.
- M. McBride and D. B. Booth, 2005, Urban impacts on physical stream conditions: effects of spatial scale, connectivity, and longitudinal trends: Journal of the American Water Resources Association, Vol. 41, No. 3, pp. 565-580.

State and Impacts: Physical Disturbance

Placeholder – includes state changes of increased ambient light, noise and heat, and relative impacts on suitable conditions for species

Placeholder - State and Impacts: Altered Food Webs

Placeholder- Pressure: Infrastructural Demands

Placeholder- State and Impacts: Water Withdrawals

State and Impacts: Wastewater - Placeholder (link to Pollution threat)

Placeholder - State and Impacts: Stormwater

Placeholder - State and Impacts: Transportation Corridors

Uncertainties and Information Gaps

The review above highlights the myriad pressures, state changes and consequent impacts associated with residential, commercial and industrial development. The growing field of urban ecology (Alberti 2008) increasingly provides information and an understanding of the distinct community-, ecosystem- and landscape-level interactions that characterize developed lands, and the unique role of humans in such systems. Changes associated with development result in species composition shifts, and changes in ecological community structure and the flows of water and materials in the Salish Sea ecosystem (e.g., Hepinstall et al. 2008, 2009). As the region becomes increasingly developed, we can expect these resultant ecological shifts to expand in extent and intensity.

Despite all that is known regarding ecological changes associated with development, significant gaps remain in quantifying the extent and relative magnitude of such impacts. A growing body of literature exists on shifts in bird, fish, and to some degree amphibian assemblages along urban-to-rural gradients in Puget Sound. Much work remains, however, to systematically investigate changes in plant communities, for which some data are available but with few syntheses, and invertebrate communities, for which little data appears to be available. Interactions between taxa, such as competition, predation and trophic relationships associated with development, have also been explored for birds and in freshwater and marine systems, but remain to be examined for other significant taxonomic groups in the Sound. A more thorough investigation of federal, state and local government reports, as well as non-governmental organization documents, may in fact provide significant information to fill many of these gaps. Such an expanded compilation of information and syntheses is thus strongly recommended

Syntheses examining biogeochemical impacts of residential, commercial and industrial development in the Salish Sea appear to be limited, particularly in the peer-reviewed journal literature. Much of the existing research on shifts in nutrient fluxes in developed landscapes such as changes in absorption and discharge rates associated with vegetation loss and increased imperviousness have come from studies in Baltimore (e.g., Groffman et al. 2002, 2003, 2004, 2005; Law et al. 2004; Pickett et al. 2008) and Phoenix (e.g., Baker et al. 2001a; Hope et al. 2005), the two urban ecosystem sites in the National Science Foundation's Long-Term

Ecological Research network. Similar comprehensive investigations remain to be compiled for the Salish Sea ecosystem. Systematic exploration of nutrient, sediment and other material loadings as a function of LULC composition and configuration within various watersheds, particularly along urban-to-rural gradients, would greatly enhance our understanding and prediction of biogeochemical trends, and resultant ecological impacts. Significant data sources exist through sampling efforts of federal (e.g., US Geological Survey), state (e.g., Washington Department of Ecology) and local (e.g., King County Department of Natural Resources and Parks) agencies. Again, some of the needed syntheses may exist in, and hence be identified through an expanded survey of, the larger body of government agency reports. However, sampling in some watersheds is limited to a single station, which is insufficient to capture the heterogeneity of landscape conditions and biogeochemical sources. Beyond data limitations, there is also the need to comprehensively analyze existing data, in order to understand the interplay between the distinct landscape characteristics of developed versus undeveloped lands. Expanded efforts at adapting existing ecosystem process models or developing new ones for the region could help us understand and predict the effects of development on biogeochemical fluxes (see the section on ecosystem models below).

Driver: Human Activities in Proximity to Shoreline

The level of human activity in the Salish Sea region both partly springs from and leads to extensive use of nearshore ecosystems. Access to shipping, fishing and other commercial and recreational endeavors makes the region an attractive location for human settlement. Expanding settlement and human activities exerts growing pressures on the ecological system. In the Driver-Pressure-State-Impacts-Response (DPSIR) conceptual model, nearshore human activities are represented as “Drivers” (Figure 3). Because shoreline modification is a consequence of these driving activities, the threat is represented as a Pressure in our review.

In the sections below, we review the Pressures of shoreline modification, and the resultant State changes and system Impacts. To avoid repetition of an existing review of this topic, we rely heavily on reviews completed by the Puget Sound Nearshore Ecosystem Restoration Project (Simenstad et al. 2009; Schlenger et al., in review) but supliment this review with other information from the peer-reviewed literature. We recommend that readers consult Simenstad et al. 2009; Schlenger et al., in review for greater details, both with respect to specific shoreline modifications and the status of distinct geographic subunits in the region. Although Schlenger et al. is currently in review and therefore not part of the peer-reviewed literature, we rely heavily on this document because the authors have essentially completed the goals of this section – to review the peer-reviewed literature of the threats associated with shoreline modification.

Given the economic and recreational impetuses leading to shoreline modification, such activities clearly can have positive impacts on human socioeconomics and well-being. However, we only focus in the current draft on the ecological changes resulting from shoreline modification rather than the benefits to human health and well-being; additionally, negative impacts of ecological changes on human health and well-being, such as decreased resource availability, impaired water quality, and increasing expenditures for shoreline restoration, are currently omitted. Lastly, Chapter 4, Effectiveness of Strategies to Protect and Restore the System, addresses the human Response to the problems associated with such modifications, and will not be covered in the present section.

Placeholder – positive and negative impacts of residential, commercial and industrial development on human health, socioeconomics and overall well-being

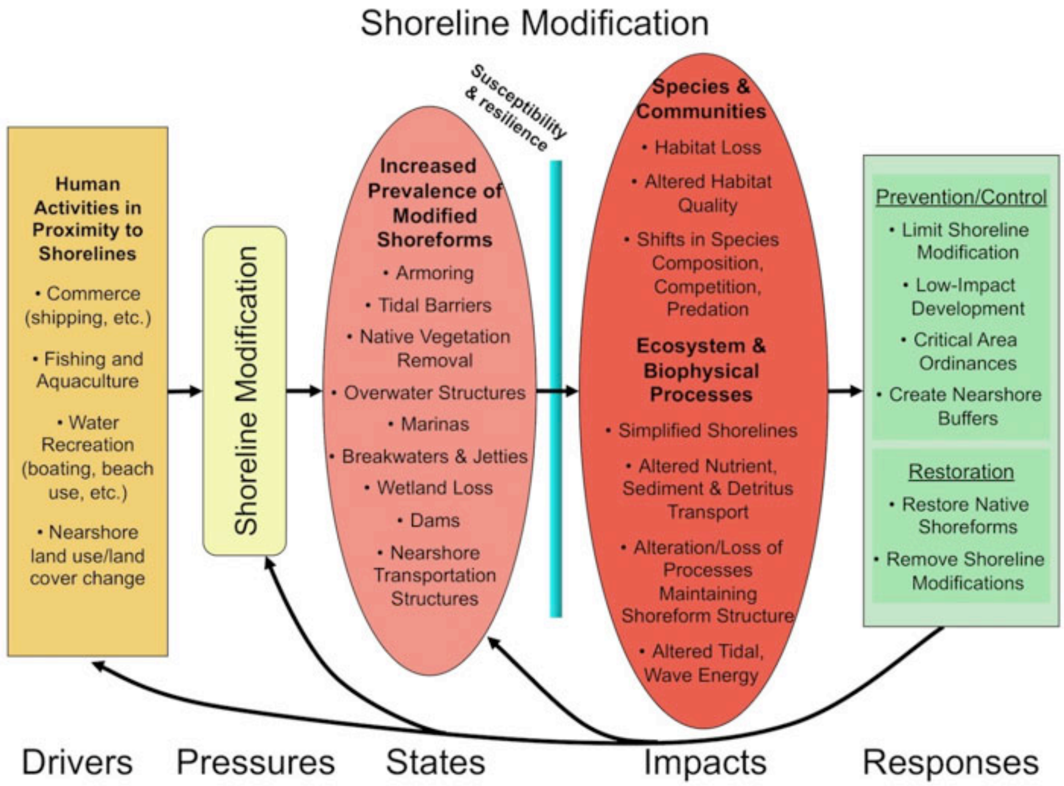

Figure 3. Driver-Pressure-State-Impacts-Response conceptual model for shoreline modification in the Salish Sea ecosystem.

1. Pressure: Shoreline Modification

Modification of shoreline regions results in a wide range of state changes in nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review; summarized in Table 3). These changes lead to impacts to the shoreline, to the adjacent upland and freshwater systems and to the Salish Sea estuary (Simenstad et al. 2009; Schlenger et al., in review; summarized in Table 4). Collectively, nearshore modification has resulted in shortening and simplification of shoreline over the past 150 years, from both direct (e.g., artificial structures) and indirect (e.g., disruption of shoreform sediment transport processes) modifications; the Sound has experienced a loss of over 1000 km of natural shoreline and the introduction of almost 400 km of artificial shoreline (Simenstad et al. 2009; Schlenger et al., in review). This loss of convoluted shoreline has resulted in an overall loss of nearshore area, leading to disruption or loss of important ecosystem functions such as sediment, detritus and nutrient transport, loss of habitat and changes in species composition.

Table 3. Extent, number and percent change in shoreline by modification type in Puget Sound, the Strait of Juan de Fuca, and southern Strait of Georgia[1]

Modification Type	Extent of Modification	Number Occurring	Percent of Modification
Armoring	1071 km	-	27%[2]
Tidal Barriers	263 km	-	7%[2]
Overwater Structures	9 km2	8972	-
Marinas	6 km2	171	-
Breakwaters & Jetties	37 km	136	-
Loss of Wetlands	273 km2	-	53% of historical extent[3]
Dams	13,000 km2 impounded	436	37% impounded[4]
Transportation Structures	383 km (312 km of roads, 71 km of railroads)	-	10%[2]

[1]Numbers derived from Simenstad et al. (2009) and Schlenger et al. (in review).

[2]Based on a total shoreline length of 3962 km.

[3]Based on historical extent of 514 km2.

[4] Based on a total sub-basin drainage area of 34,710 km2.

Table 4. Summary of direct (D) and indirect (I) impacts to nearshore processes by shoreline modification type[1]

	Modification Type							
Nearshore Processes Impacted by Shoreline Modification	Armoring	Tidal Barriers	Native Vegetation Removal	Overwater Structures	Marinas	Breakwaters & Jetties	Dams	Transportation Structures
Sediment Input	D		I	I	I	I	D	D
Sediment Transport	D	D		I	D	D	D	I
Erosion/Accretion of Sediment	D	D	I	I	D	D	I	D
Tidal Flow	I	D			I			D

Tide Channel Formation and Maintenance	I	D			I			I
Distributary Channel Migration	D	D	I			I	D	I
Freshwater Input	I		I		I		D	I
Detritus Import and Export	D	D	D	I	I	I	I	D
Exchange of Aquatic Organisms	D	D		D	D	D	D	D
Physical Disturbance	D	I		I	D	D	I	D
Solar Incidence	I		D	D	D	I		I

[1]Partially reproduced from Schlenger et al. (in review), Table 4-19, pg. 123.

Many shoreline modifications are for residential, commercial and industrial purposes. Nearshore ecosystems are thus subject to the same pressures, state changes and impacts associated more generally with development-related LULC changes, such as altered material and water fluxes due to increased imperviousness (Schlenger et al., in review). However, specific modes of shoreline modification have distinct characteristics with respect to their impacts on nearshore environments. We review here many of these various state changes and their associated impacts.

State: Increased Armoring

Shoreline armoring refers to structures largely aimed at erosion control from coastal wave movement, and for retention of fill zones. Such armoring consists of walls or bulkheads, constructed of rock or concrete, erected parallel to shorelines. Covering over 1070 km of Puget Sound shorelines (Schlenger et al., in review), armoring is particularly prevalent in highly developed residential, urban or industrial centers, due to a combination of the need to protect developed structures (e.g., roads, buildings) and the increased potential for erosion due to the removal of vegetation for land development (Alberti 2008; Schlenger et al., in review). For instance, armoring frequently co-occurs with nearshore roads, railroad passages, and/or other transportation infrastructure (Simenstad et al. 2009; Schlenger et al., in review). Of the various shoreline modification forms, armoring is the most common, comprising 74 percent of all artificial shoreforms (Simenstad et al. 2009).

Impacts:

Armoring significantly alters the movement of sediments and debris that provide physical structure to beaches and other nearshore zones (Simenstad et al. 2009; Schlenger et al., in review). By design, armoring structures block natural, more gradual upland erosion processes that deliver sediments and replenish shoreline materials carried away by waves and tides. In place of such processes, the abrupt physical barrier serves to intensify waterward erosion of waves, further altering beach structure.

These changes to movements of sediment and debris are one of the primary impacts leading to degradation of river deltas within the Salish Sea ecosystem. Approximately 44 percent of river delta extent (188 km2 of the 427 km2 historical area) has been lost due to impacts such as armoring (Schlenger et al., in review). Shoreline modifications such as armoring alter both the transport of sediments into river deltas and the distribution of sediments within the delta itself (Miles et al. 2001; Johannessen and MacLennan 2007). In turn, degradation of river deltas has significant ecosystem impacts, including loss of habitat and restriction of species ranges (e.g., salmon and other fish, shorebirds and the benthic invertebrates they depend on) (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008). Resultant changes in sediment flows also increases estuarine turbidity.

Armoring results in the degradation of bluff-backed and barrier beaches (Canning and Shipman 1995; Johannessen and MacLennan 2007), particularly in South Central Puget Sound (Schlenger et al., in review). Bluff-backed beaches have declined by approximately 8 percent from their historical extent due to a range of factors including armoring (Simenstad et al. 2009; Schlenger et al., in review). Approximately 33 percent bluff beaches include some level of armoring, leading to disruption of the sediment and debris transport process that feeds these and nearby down-drift beaches. Coastal bluffs provide an estimated 90 percent of sediment to beaches along the Sound (Downing 1983), which in turn affects resilience of coastal embayments that depend on this input. Barrier beaches, which serve as protection for estuary lagoons and other coastal embayments, have also declined by 12 percent of their historical extent; of these, 27 percent include shoreline armoring (Simenstad et al. 2009; Schlenger et al., in review). Degradation and loss of bluff and barrier beaches result in loss of invertebrate habitats (Sobocinski 2003; Dugan and Hubbard 2006; see Schlenger et al., in review), which impacts fish, mammals and birds that feed on them. Armoring these systems also results in loss or impairment of spawning habitat of forage fish such as surf smelt and sand lance (Rice 2006; Penttila 2007) and herring, which may lead to declines in some species that feed upon these fish or their eggs (surf scoter populations, for instance Anderson et al. 2009).

Changes in sediment transport due to armoring have also contributed to loss or fragmentation of coastal embayments, such as inlets, barrier estuaries, barrier lagoons, closed lagoons and marshes (Schlenger et al., in review). Compared with historical occurrence, 53 of 173 open coastal inlets, 84 of 240 barrier estuaries, and 89 of 222 barrier lagoons have been lost. Closed forms of coastal embayments, such as lagoons and marshes that do not interface with open estuary, exhibit similar trends: comprising approximately 1.6 km of the Puget Sound shoreline (down from a historical extent of 2.6 km), only about 81 of 249 historic closed lagoons and marshes remain. As noted above, coastal sediment transport processes that create and maintain structure for barrier beaches form the boundaries for coastal embayments; disruption of such transport due to armoring in turns leads to the degradation of embayments (Schlenger et al., in review). Losses of embayments have been noted to have significant impact on juvenile Pacific salmon that use these habitats for feeding (Beamer et al. 2003; Fresh 2006). Other significant impacts include altered nutrient inputs and overall water quality, loss of or diminished primary productivity, and loss of biodiversity (Schlenger et al., in review).

Placeholder – discussion of riprap impacts on aggregating some fish species, and increasing velocity along river banks

State: Construction of Tidal Barriers

Tidal barriers consist of structures such as dikes, levees and tide gates that are used to restrict or divert tidal flows. They are often used to block tide waters (or in the case of tide gates, to drain water) from delta regions that have been converted to agricultural lands (Schlenger et al., in review). Tidal barriers are typically constructed of large rock and other heavy materials to prevent damage from flood waters. According to shoreform database estimates, approximately 418 km of tidal barriers exist within Puget Sound nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

Because of the nature of their construction and use, tidal barriers have particularly significant impacts on river deltas (Schlenger et al., in review). As with armoring, these barriers alter the transport and distribution of sediments to and within deltas, coastal marshes and tidal channels (Thom 1992; Barrett and Niering 1993; Brockmeyer et al. 1997; Bryant and Chabreck 1998; Hood 2004). These impacts in turn alter the formation and maintenance of tidal flow channels, and hence the overall structural integrity of river deltas. Changes in sedimentation also have potentially negative impacts on eelgrass and kelp survival (Mumford 2007; Schlenger et al., in review). As a consequence, shorebirds, fish and benthic invertebrates that rely on such river delta vegetation for foraging, spawning and refuge habitat experience declines in their abundance and distribution (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008; cited in Schlenger et al., in review). Turbidity in the vicinity of river mouths also increases.

Placeholder – information on number of deltas with >75% coverage by tidal barriers, and the number being restored to remove tidal barriers

Open and closed coastal embayments are also significantly impacted by tidal barriers (Schlenger et al., in review). Barriers occur within the immediate vicinity of 16 percent of open coastal inlets and 21 percent barrier estuaries in the Sound (Simenstad et al. 2009). The structure of embayments, whose boundaries are dependent on persistent replenishment of sediments from both tidal and more upland flows, is frequently modified by the changes in sediment transport induced by tidal barriers (Schlenger et al., in review). Such shifts particularly alter or disrupt the morphology and vegetation composition of nearshore marshes (Barrett and Niering 1993; Bryant and Chabreck 1998; Hood 2004), and limit the availability of detrital nutrients used by aquatic organisms (Schlenger et al., in review).

State: Native Vegetation Removal

Changes in land cover, particularly removal and/or fragmentation of native vegetation, is frequently associated with artificial shoreline modifications (Schlenger et al., in review). Residential and industrial development, and the changes in land cover that it entails, is prevalent along Puget Sound shorelines, particularly in the central and southern Sound regions (Alberti et al. 2004; Simenstad et al. 2009) .

Impacts:

As described above (see “State: Altered Biogeochemistry and Hydrology” under “Pressure: Land Use/Land Cover Change”), changes in land use and land cover modify the rates and volumes of upland water and material fluxes (Tague and Band 2001; Booth et al. 2002; Wickham et al. 2002; Brett et al. 2005; Kaye et al. 2006; Cuo et al. 2009), which in turn translate into altered transport into nearshore ecosystems.

Changes in sediment, water and nutrient fluxes due to upland vegetation conversion alter the geomorphic structure and ecosystem functioning of nearshore ecosystems (Schlenger et al., in review). Changes in upland transport of sediments interact with in-water fluxes to modify the structure and stability of shore banks, beaches and embayments. Degraded structural and biogeochemical changes to embayments and river deltas in turn alter, and often simplify, food webs and communities that depend on these shoreforms for shelter and foraging habitat (Griggs 2005; Buchanan 2006; Dethier 2006; Fresh 2006; Mumford 2007; Tonnes 2008; cited in Schlenger et al., in review).

State: Construction of Overwater Structures

Overwater structures comprise a general class of shoreline modification that includes fixed and floating docks, fixed piers, bridges, floating breakwaters, moored vessels, and support and stabilization piles. Approximately 6927 overwater structures can be found in the Puget Sound region, comprising a total area of approximately 6.5 km2 (Simenstad et al. 2009; Schlenger et al. in review). The severity of nearshore impacts of a given overwater structure depend on some of the following physical characteristics that determine its physical profile in and above the water (Nightengale and Simenstad 2001; Schlenger et al., in review): the structure’s size and shape; its height above the water and the depth of water below it; the number of support pilings it requires; its orientation to and location along the shore; and its proximity to other overwater structures.

Impacts:

One of the key impacts of overwater structures is shading of nearshore habitats (Nightengale and Simenstad 2001; Schlenger et al., in review). Aside from the obvious implications for nearshore plants (Dennison 1987; Kenworthy and Haunert 1991), shading also impacts the distribution, behavior and survival of fish and other aquatic wildlife that occupy adjacent shoreline habitats. Sharp gradients of light and shadow, such as those that occur near overwater structures, affect feeding behavior and efficiency of visual foragers (e.g., salmon, Dungeness crab) as well as fish schooling and migratory movements (Nightengale and Simenstad 2001; Scheuerell and Schindler 2003; Thom et al. 2006; Schlenger et al., in review).

Placeholder – discussion of overwater structure impacts on fish aggregation vs. deterrence (e.g., does the shade help keep water temperatures cooler?)

As with other shoreline modifications that pose physical barriers, structural support pilings interfere with tidal flows and wave movements (Nightengale and Simenstad 2001; Schlenger et al., in review). Individual pilings may have negligible impacts on water movements and energy, depending on their size. However, because structures typically have multiple rows of pilings, these supports have cumulative impacts that attenuate wave energy, with consequent shifts in the deposition and distribution of adjacent and downdrift shoreline sediments.

Also associated with overwater structures, particularly those of older construction, is the potential introductions of contaminants into nearshore waters (Poston 2001; Schlenger et al., in review). Older, creosote- or copper-treated wood structures have been demonstrated to leach polycyclic aromatic hydrocarbons and copper arsenate compounds, respectively, into aquatic ecosystems (Valle et al. 2007).

Placeholder – discussion of short-term effects during construction with pile driving and sediment disturbance, particularly with respect to timing of construction relative to migrating animals

State: Construction of Marinas

Marinas are comprised of a diversity of in-water and/or overwater structures, as well as adjacent nearshore modifications such as parking lots and service buildings, that vary in impact depending on their specific physical characteristics (Schlenger et al., in review). Building structures of varying size, shape and orientation in conjunction with water vessel moorings alter both the geomorphic characteristics of shorelines and the flows of water and sediments; accompanying breakwaters and jetties (see below) further exacerbate impacts. Approximately 0.3 percent (around 6 km2) of Puget Sound shoreline is covered by over 170 marinas, with about one third occurring in the south Sound (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

The impacts of marinas are significant on beach systems, river deltas and coastal embayments (Schlenger et al., in review). Physical in-water and overwater barriers associated with marinas alter or disrupt the transport of sediment, coarse debris and detritus, thereby degrading beach structure immediately adjacent to as well as downdrift from the marina. As noted above, shading from accompanying overwater structures also impacts plant productivity and aquatic wildlife foraging and movement behavior (Nightengale and Simenstad 2001; Schlenger et al., in review). Marinas constructed near river deltas or coastal embayments similarly alter both upland and in-water sediment transport processes that maintain the structure and water and material flows within these shoreforms. Upland armoring often accompanies marinas, degrading nearshore habitats for wildlife and further disrupting land-water interactions (Simenstad et al. 2009; see "State: Increased Armoring" above).

Marinas also introduce chemical contaminants into nearshore ecosystems (Poston 2001; Schlenger et al., in review). As with overwater structures, leaching of chemicals from treated wood structures is a potential risk. Perhaps more significant and prevalent, however, are risks of contaminants released into water and sediments from moored vessels and upland parking facilities. These petroleum-based and other forms of contaminants have significant impacts on plants, aquatic and nearshore wildlife and general nearshore food web structure (Schlenger et al., in review).

Placeholder – discussion of impacts from tin-based antifouling paints that are stored in the bottom sediments, from when these paints were legal in the USA

Placeholder – impacts of noise pollution from vessel traffic and industrial activity in and around marinas

Placeholder – potential impacts from stray electrical currents from marinas

Placeholder – positive vs. negative impacts on wintering populations of birds

State: Construction of Breakwaters and Jetties

Similar to tidal barriers (see "State: Construction of Tidal Barriers" above), breakwaters and jetties are structures designed to dissipate wave movement and energy, particularly near harbors, marinas and areas where vessels are moored (Schlenger et al., in review). Some breakwaters and jetties are composed of heavy rock or concrete armoring, while others are comprised of free-floating or anchored structures. There are 136 recorded breakwaters and jetties in the Salish Sea ecosystem, with about 65 percent occurring in the northern portion (Simenstad et al. 2009; Schlenger et al., in review). They range in length from as little as 5 m to as long as 5 km (Schlenger et al., in review).

Impacts:

Impacts of breakwaters and jetties generally depend on their orientation to the shoreline (Schlenger et al., in review). Structures oriented parallel to the shore lead to deposition of sediment on the waveward side, resulting in accretion beaches and a deepening of shoreline channels on the opposite side of the structure. Breakwaters and jetties that are perpendicularly oriented disrupt shoreline sediment and detritus transport processes that maintain the geomorphology of downdrift beaches and coastal embayment boundaries. Breakwaters and jetties erected adjacent to river deltas and coastal embayments also serve to disconnect these aquatic ecosystems from the broader Sound and from one another. The resultant changes in shoreform morphology, connectivity and nutrient and water flows leads to degraded habitat quality for nearshore wildlife and plant communities (Schlenger et al., in review).

Placeholder – potentially positive impacts of breakwaters and jetties providing shelter for wintering bird populations in storms

State: Loss of Wetlands

Through a variety of forms of shoreline modification – particularly armoring and tidal barrier impacts on river deltas and coastal embayments as well as outright filling – significant loss of wetlands has occurred or is occurring along Puget Sound nearshore ecosystems (Simenstad et al. 2009; Schlenger et al., in review). Approximately 53 percent, or 273 km2 out of 514 km2, of historical wetland extent has been lost to these various stressors. Of particular concern are losses of tidal freshwater and oligohaline transitional wetlands: these two wetland classes have lost approximately 93% of the historical extent (Schlenger et al., in review).

Impacts:

Losses of these important coastal ecosystems have significant implications. Ecosystem functions performed by wetlands, such as food and nutrient production, contaminant filtration, breeding and feeding habitat provision, become considerably impaired as wetland area diminishes (Schlenger et al., in review). Wetland losses particularly impact Chinook populations, since these shoreforms provide significant habitat during juvenile growth stages (Bottom et al. 2005; Fresh 2006).

Placeholder – expand discussion of impacts on Chinook

Placeholder – expand overall discussion of the ecological importance of the loss of wetlands

State: Construction of Dams

The number and distribution of dams in the Salish Sea ecosystem is of significant concern in terms of their impacts, which vary as a function of a given dam's position in the watershed and the number of other dams up- and downstream of it (Neuman et al. 2009; Simenstad et al. 2009; Schlenger et al., in review). A total of 436 dams can be found in the Puget Sound basin, impounding approximately 13,000 km2, or 37 percent, of the total sub-basin drainage area (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

By diverting or constraining the flow of water, sediments, nutrients and organic matter, dams prevent transport of materials necessary for the persistence of downstream nearshore ecosystems, particularly in river deltas and coastal embayments (Schlenger et al., in review). Along with upland sources, rivers and streams deliver sediments and organic matter that provide structural integrity to nearshore ecosystems, replenishing materials that are washed away via tides and waves. These materials, as well as nutrient and freshwater inputs, are important for the persistence of downstream plant (e.g., kelp, eelgrass) and animal (e.g., shellfish, juvenile salmon) populations and food web interactions (Schlenger et al., in review). Changes in water flow rates and levels result in water temperature regime changes both in upstream riverine and downstream nearshore ecosystems (Schlenger et al., in review). Significant disruption of native vegetation, soils and hydrologic regimes also occurs in reservoirs upstream of the dams, impacting upland biota and ecosystem functions in ways that then further impact downstream nearshore systems (Schlenger et al., in review).

Placeholder – expanded discussion of dam impacts; include specific discussion of the effects of dams on connectivity, stream temperature, migratory fish, and the timing and levels of flows

State: Construction of Transportation Structures

A number of different classes of transportation structures are found within close proximity to nearshore ecosystems, including railroads, nearshore roads, and stream crossings (Simenstad et al. 2009; Schlenger et al., in review). Roads and railroads occur along 312 and 71 km of Puget Sound shoreline, respectively, comprising almost 10 percent of its total length (Simenstad et al. 2009; Schlenger et al., in review).

Impacts:

Impacts of these features are analogous to and compounded by the effects of upland impervious surfaces, particularly with respect to changes in hydrology and biogeochemistry and increased contaminant runoff (see "State: Increased Imperviousness" under "Pressure: Land Use/Land Cover Change"). Nearshore transportation corridors and structures contribute to disruptions in upland replenishment of sediment and detritus to beach and embayment shoreforms, particularly through interactive impacts with other shoreline modifications (e.g., armoring, vegetation removal, etc.). Fill material used to bolster transportation routes further alters the geomorphic structure of, and often eliminates, shoreline ecosystems (Schlenger et al., in review). Construction of transportation corridors frequently disrupts connectivity within and among shoreline ecosystems, particularly in the form of overpasses through or over river deltas and embayments. Lastly, increased contaminant loadings occur as a result of nearshore transportation structures, both directly deposited by automobiles and trains and indirectly mobilized via surface water runoff across impervious surfaces (Booth et al. 2002, 2004; Kaye et al. 2006; Krahn et al. 2007; Schlenger et al., in review).

States and Impacts: Cumulative Effects of Shoreline Modifications

As illustrated above, most of the various forms of shoreline modification have comparable impacts on nearshore ecosystems (Schlenger et al., in review): disruption of sediment and detrital transport rates, levels and mechanisms; altered and often simplified estuarine and freshwater flow pathways; increased contaminant levels; and general disruption of nearshore ecosystem functions and resultant habitat degradation. Exacerbating the effects of shoreline modification is the fact that often several of these modification forms co-occur within a given location. In change assessments for the Puget Sound, Strait of Juan de Fuca and Strait of Georgia Basins, 65 percent of drainage catchments include more than one type of modification (Simenstad et al. 2009; Schlenger et al., in review). For example, armoring commonly co-occurs with other stressor types, most frequently accompanying nearshore roads (in 46 percent of catchments). These findings suggest a significant risk of cumulative, synergistic impacts from multiple stressors.

Placeholder – expanded discussion of and citations for cumulative shoreline modification impacts

Uncertainties and Information Gaps

The uncertainties and knowledge gaps associated with shoreline modification in the Salish Sea ecosystem reflect questions in data availability and quality. In addition to extensively reviewing the forms of shoreline modification and their impacts, the PSNERP Strategic Needs Assessment Report (Schlenger et al., in review) also discuss such uncertainties in detail; we thus present only an overview of this topic.

One source of uncertainty lies in the quality of datasets available for analyzing shoreline modification extent and impacts. A comprehensive analysis covering the extent of shoreline in Salish Sea required compilation of data sets from a variety of sources, each of which includes its own level of accuracy and uncertainty. Inaccuracies are potentially most problematic for historical conditions, for which data are limited at best and require estimating of sedimentation rates and other key shoreline formation processes. Such inaccuracies can of course affect change detection and estimates, but are unavoidable and must therefore simply be taken into consideration as fully as possible.

In addition to those entailed in geographic measurements of the extent of shoreforms and their modification, uncertainties exist in the linkages between state changes and their systemic impacts. Assessment of impacts in Schlenger et al. (in review) and Simenstad et al. (2009) were based on review and synthesis of empirical investigations in peer-reviewed and gray literature. As reflected in our review above, such synthesis provides a qualitative understanding of potential impacts to Salish Sea biota and ecosystem processes; investigations targeted at specific cause-and-effect linkages are necessary to quantify the level of impacts. At the same time, applicability and generalizability of targeted studies to the broader system requires systematic review and evaluation. This requirement is particularly necessary when drawing conclusions from studies that examine causal linkages between shoreform modification and ecological impacts in systems comparable to, but not within, the Salish Sea region.

Lastly, specific scales of analysis may result in biases and uncertainties in estimated state changes and their impacts. PSNERP's assessments of shoreline modification were aggregated at the catchment level as the finest scale of measurement (Simenstad et al. 2009; Schlenger et al., in review). Because such catchments vary in size throughout the region, measures of the extent of shoreline modification that are aggregated to this level can over- or underestimate absolute levels and intensity of modification within a given segment of the watershed. Schlenger et al. (in review) note that refined, more detailed site-level assessments can correct for these uncertainties. Additionally, some level of aggregation – preferably at fine enough scales to capture key biophysical processes such as sediment transport rates (as is true for catchments) – is all but necessary for broader-scale, relative trends that characterize segments of the Salish Sea ecosystem.

Driver: Pollution in the Puget Sound Basin

In its broadest sense pollution is often thought of as the introduction of unwanted or undesirable substances or conditions into the natural environment. Virtually all pollution types described in this section are unintended consequences of the daily activities of humans – diving cars, heating homes, growing food, building shelter, generating waste, manufacturing goods and so on. A Driver-Pressure-State-Impacts-Response (DPSIR) conceptual model is used here to help organize the complex information that describes these human activities and the pressures they create on the ecosystem (i.e. "Threats"). In addition it can provide context for discussing pollution-harm in the ecosystem and to humans, and the range of possible strategies we might employ to mitigate the threat (Figure 4).

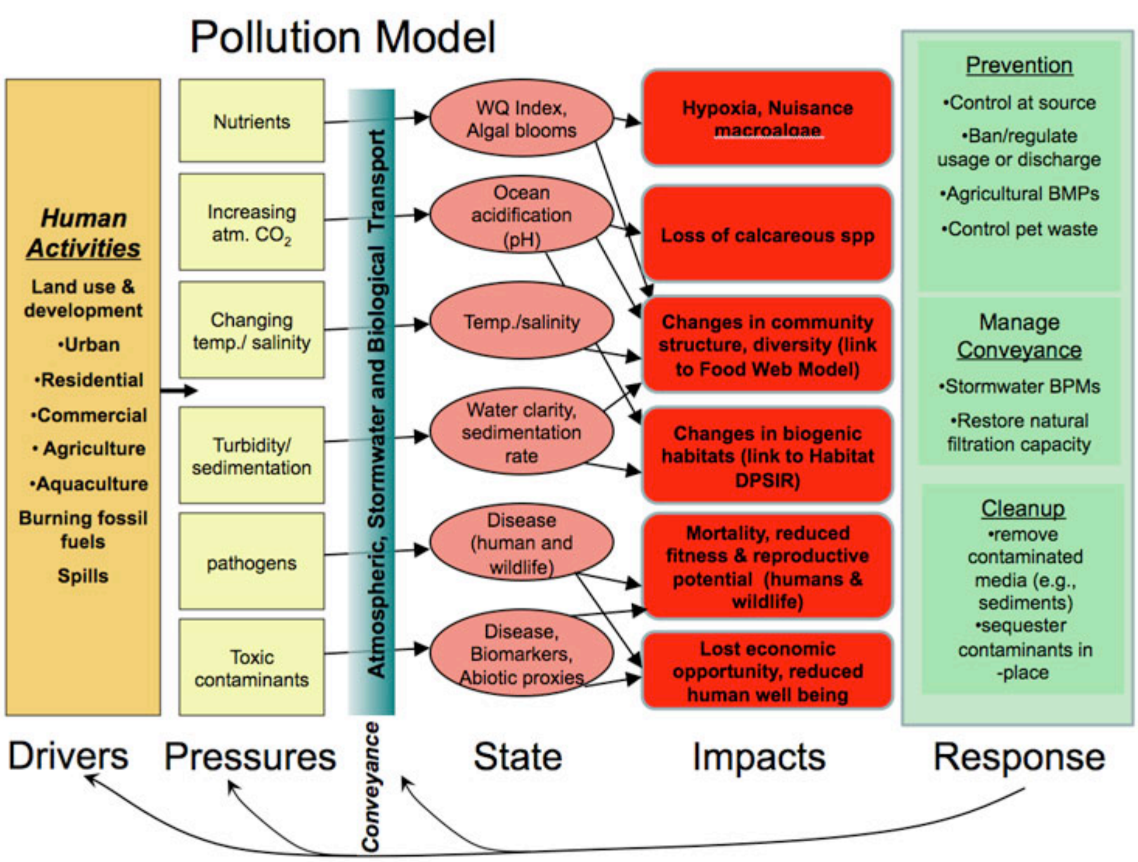

Figure 4. Driver-Pressure-State-Impacts-Response conceptual model for Pollution in the Salish Sea ecosystem.

Human activities that generate pollution pressures can be organized according to the types of land use that generate characteristic pollution types, including urban, residential, commercial, agricultural activities. These can be overlain with cross-cutting activities (such burning fossil fuels) that occur virtually everywhere. Additionally, spills of chemicals, nutrients, soils, sediments or other unintentional episodic introductions of pollutants can cut across land use patterns.

Pollution occurs when human activities (a) generate toxic chemicals, (b) concentrate or make available naturally occurring substances to levels that can be harmful, (c) change conventional water quality characteristics (e.g., temperature) or (d) introduce disease pathogens or conditions that exacerbate diseases. In many cases pollutants may be generated or manufactured or released in one place and then transported to other areas where humans or biota in the ecosystem can be exposed to the pollutant. It is as important to understand these naturally occurring conveyance pathways such as stormwater, groundwater, air movement, and biological transport of pollutants because these are the mechanisms whereby pollutants move from their source to where they cause harm in the environment. In particular the degree to which stormwater or surface runoff patterns have been altered by human activities helps us understand how our actions may exacerbate or mitigate movement of pollutants in the environment.

The degree of potential harm or toxicity of the pollutant is related to the amount of the pollutant loaded to the system (the dose), the degree to which pollutants are subsequently concentrated in the environment, the fate and the sensitivity of the organism or ecosystem processes that are affected, and their ability to recover once the pressure is reduced (resiliency).

Although State, Impacts and Response components of this model are treated in detail in separate chapters of this Science Update, most definitions of Threat include some indication of harm to living organisms or ecosystem processes. Hence we include in this Threats Chapter some examples of harm related to pollution pressures, with greater detail on state and impact presented in Chapter 2a , Biophysical status of Puget Sound.

Pressure: Nutrients - Placeholder

Pressure: Increasing Atmospheric Carbon Dioxide - Placeholder

Pressure: Changing Temperature and Salinity - Placeholder

Pressure: Turbidity and Sedimentation - Placeholder

Pressure: Pathogens and Disease - Placeholder

Pressure: Toxic Contaminants in Puget Sound

The threat of pollution pressures in the Puget Sound Basin depends on where, when, amount, and type of contaminants that are loaded to the system (Figure 5). This section focuses on Washington's inland marine and estuarine waters including Puget Sound's main basins, Hood Canal, eastern Strait of Juan de Fuca, San Juan archipelago, and southern Strait of Georgia, (hereafter collectively referred to as Puget Sound), and the conveyance-pathways to that marine/estuarine system. Subsequent contributions to this Chapter will review toxic contaminants in freshwater systems, including the lakes, rivers, streams, wetlands and groundwaters that drain to Puget Sound or the Pacific Ocean.

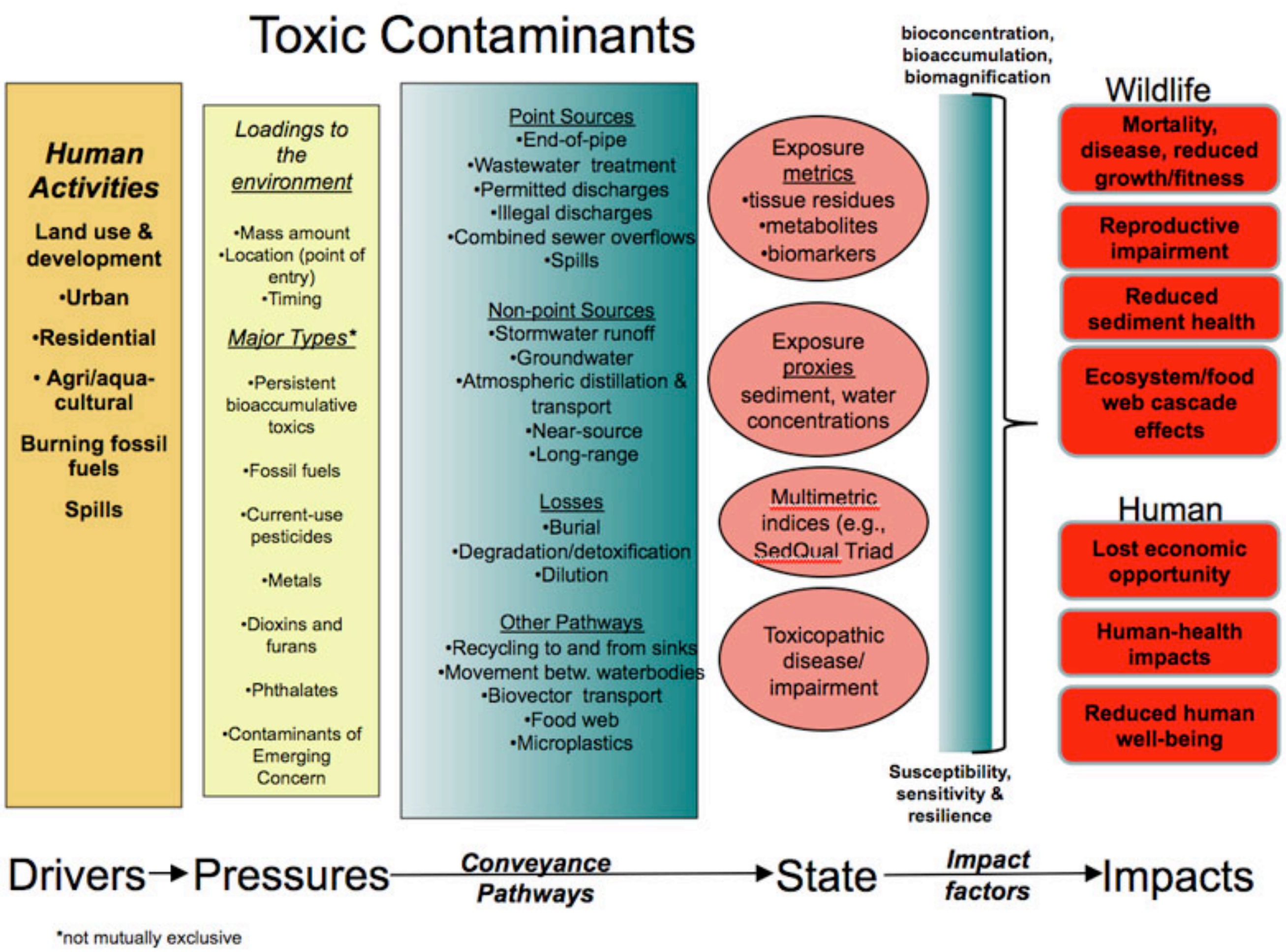

Figure 5. Driver-Pressure-State-Impacts-Response conceptual model for Toxic Contaminants in the Salish Sea ecosystem.

Puget Sound's fjord-like physiography, oceanographic isolation of some of its major basins, and relatively long water residence time may increase the susceptibility of its biota to contamination (Thomson 1994). Because the Sound possesses such a wide range of oceanographic conditions and habitats it also enables species that range from fully marine to diadromous to complete their entire life cycle within its waters, potentially exposing sensitive life stages to contamination.

Loading of Toxics to Puget Sound

The degree to which biota in the Puget Sound ecosystem are exposed to toxic contaminants depends on a complex interaction among the human activities that create the chemicals (e.g., land use, spills, burning fossil fuels), amounts and types of chemicals produced, and how they are conveyed to the ecosystem (Figure 5). Washington Department of Ecology (2010) have conducted or are currently conducting, sponsoring, or facilitating twenty studies designed to quantify loadings to and support the control of toxics in Puget Sound. These include inventories of chemicals of concern, estimates of chemical loadings to Puget Sound and the land-use activities that produce the chemicals, models of how chemicals move through the system, and evaluation of the fate and transport of these chemicals in the biological component of the ecosystem. Lubliner (2007) described some of these complex interactions within the context of estimating the total maximum daily load of chemicals to a water body.

Chemicals of Concern

Deciding which chemicals to evaluate is a daunting challenge: of the more than 53 million substances inventoried in the American Chemical Society's Chemicals Abstract Service , over 100,000 have been registered for use in commerce in the USA. Only a relatively few have undergone much scrutiny or are regularly measured in the environment (Muir and Howard 2006). Their sheer numbers necessitate a scheme to select indicator chemicals that represent a wide range of chemical types.

The Washington Department of Ecology selected 17 Chemicals of Concern on which to focus evaluation of loadings to Puget Sound (Hart Crowser 2007). Selection criteria were based on concern for threats to biota or humans, chemicals that represent a broad range of conveyance pathways, and for which some monitoring data exist. This list includes a broad range of toxic contaminants that can be organized into logical groupings including metals (arsenic, cadmium, copper, lead, mercury and zinc); persistent bioaccumulative toxics (polychlorinated biphenyl ethers or PCBs, brominated flame retardants, or polybrominated diphenyl ethers [PBDEs], and chlorinated pesticides such as dichlordiphenyltrichloroethane and its metabolites [DDTs]); fossil fuels or their derivatives including polycyclic aromatic hydrocarbons (PAHs), oils and greases; one plasticizer (phthalates); nonylphenol, a suspected endocrine disrupting compound (EDC) and the herbicide triclopyr. Many of these pollutants are routinely measured by large-scale monitoring programs such as the national Mussel Watch Program (Kimbrough et al. 2008), and the Puget Sound Assessment and Monitoring Program for sediments (Dutch et al. 2009) and fish tissue (West et al. 2001), as well as regional monitoring programs such as King County's marine water and sediment monitoring (King County 2010).

Chemicals of Emerging Concern (CEC)

CEC is a widely used term to categorize new environmental contaminants, as well as those that may have existed for some time, but whose threat is only now becoming known. Some CECs were included on Department of Ecology's Chemicals of Concern list (Hart Crowser 2007), such as nonylphenol and bis(2-ethylhexyl) phthalate; others that are commonly discussed as threats include bisphenol-A, synthetic estrogen, and perfluorinated compounds, some of which are found in commercial goods, or may originate from the wide range of chemicals in pharmaceuticals and personal care products. Lubliner et al. (2010) measured 172 organic compounds including 72 pharmaceuticals and personal care products from three wastewater treatment plants that discharge to Puget Sound, and characterized the degree to which these chemicals are removed from wastewater or biosolids by various enhancements to secondary treatment. A number of these compounds exhibit endocrine disrupting properties, and are the focus of intense ecotoxicological research worldwide (Sumpter and Johnson 2005).

Synthetic polymers, or plastics, in the environment are a unique category of CEC because they not only pose multiple disparate threats to the ecosystem but also a unique conveyance mechanism for toxic chemicals from water to biota. Wildlife can be entangled by litter or harmed by ingestion of plastic debris, alien species can attach to and be transported by drifting litter, and benthic organisms be smothered by accumulation of plastics (see reviews in Derraik 2002, Moore 2008). In addition, the plastic itself can be toxic, and it can exacerbate exposure of organisms to other toxics. Plastic microparticles (<5mm) are created in the environment by degradation of larger litter (Thompson et al. 2004), or by the unintentional or intentional release of industrial microplastic stock. These particles can adsorb and concentrate contaminants from marine waters, including a number of toxics described earlier (Mato et al. 2000). Such particles can be subsequently ingested by a wide variety of marine organisms, thereby exposing consumers and creating a point of entry for water-column toxics to the food chain.

Conveyance Pathways of Toxics to Puget Sound

Hart Crowser (2007) cataloged nine important pathways or sources of pollutants to Puget Sound, many of which apply to freshwater systems as well:

- Aerial transport – aerial contaminants can be deposited or recondensed to terrestrial or aquatic surfaces. These pollutants include not only direct inputs to the atmosphere from human activities (e.g., from driving cars) but also those already in the environment that may be evaporated, distilled or fractionated, and transported via atmospheric processes. (e.g., Simonich and Hites 1995).
- Surface runoff – wherein stormwater carries terrestrially originating pollutants to receiving waters. Can be exacerbated by impervious surfaces (e.g., Lubliner 2007).
- Groundwater discharge – wherein subsurface groundwaters carry pollutants to receiving waters
- Discharges from industrial and municipal wastewater treatment plants (e.g., Lubliner 2010),
- Discharges from combined sewer overflows
- Direct spills (e.g., oil) to the system

Transport of pollutants in and out of Puget Sound by exchange with oceanic waters

- Reintroduction of pollutants leached, resuspended, or concentrated into biota from contaminated sediments
- Biological transport of pollutants (e.g., Ewald et al. 1998)

Surface runoff or stormwater is the primary conveyance for many toxic contaminants of concern in Puget Sound, and the ultimate source for the bulk of these toxics has been attributed to everyday activities of people in developed residential areas, rather than industrial or municipal discharges (EnviroVision et al. 2008). Pollution from runoff is the sum of contamination from many diffuse, "non-point," sources. As such it is difficult to characterize, evaluate or control. The PEW Oceans Commission (2003) characterized non-point source pollution as "…the greatest pollution threat to our oceans and coasts… the situation requires that we apply new thinking about the connection between the land and the sea, and the role watersheds play in providing habitat and reducing pollution."

Point source releases such as a discharge pipe release monitored and known amount of contaminants into receiving waters. The National Pollution Discharge Elimination System (NPDES) is designed to control pollutants at such point sources to protect water quality for drinking, fishing, swimming and other activities. All discharges to waters of the State must have an NPDES permit, which includes municipal and industrial wastewater, stormwater from certain jurisdictions, and general permits to cover a variety of other activities.

A large oil or other chemical spill poses a singular and significant threat to Puget Sound. Over 20 billion gallons of oil and other toxic chemicals are transported through Washington State by various means annually (Jensen 2009). Schmidt-Etkin (2009) reported the greatest potential risks of a worst-case oil spill in Puget Sound come from oil tankers, cargo vessels and oil barges. The largest oil vessels entering Puget Sound can carry up to 35,000,000 gallons of oil (OSAC 2009). Although the probability of a large oil spill from these vessels is relatively low, a large spill could have devastating, long-term impacts to natural and cultural resources in Puget Sound. Washington state efforts relating to oil or other chemical spills are focused on spill prevention, preparedness and response.

Losses/removal of toxic contaminants from the ecosystem - Placeholder

- Burial
- Degradation/detoxification
- Dilution/mixing
- Biological transport

Other Pathways (placeholder)

- Recycling to and from sediments
- Movement between water bodies
- Biological transport
- Trophic transfer (e.g., biomagnification)

1. State and Impact in the Ecosystem

By its definition, threat implies harm to biota, humans, or ecosystem function. The Toxic Contaminants DPSIR conceptual model helps to link the threat from human activities, contaminants sources, loadings, and conveyance pathways to the states of ecosystem health that are of concern (Figure 5). Contaminant states can be measured in biota as exposure, or concentration of contaminant residues in tissues, presence of contaminant metabolites or toxicopathic disease. Contamination of sediments and water are also often measured as a proxy for biota-exposure, based on known or surmised bioconcentration or bioaccumulation factors (e.g., see Johnson et al 2002).

Sediment Health - placeholder

- Sediment quality triad, a unique multimetric index of sediment quality that combines toxic contaminants, toxicity, and infaunal community characteristics

Biota Health

Once released into the environment, many chemicals of concern can persist for long periods of time and contaminate extensive areas. Chapter 2a summarizes major aspects of the distribution of toxic contaminants in Puget Sound's abiotic media (primarily sediments) including a the sediment quality triad, a multimetric evaluation of sediment quality related to toxic contamination. The degree to which biota are threatened by toxic contamination relates to all the complexities described in the Driver-Pressure-Conveyance above, combined with the susceptibility and sensitivity of organisms to exposure, the fate and transport of toxics in the environment and in the food web, the degree to which chemicals accumulate in tissues or are metabolized, and how resilient biota are once the pressure is removed.

Impacts to biota can be measured as direct health impairments to individuals e.g., mortality, immunosuppression, reduced fitness, or reproductive impairment that may ultimately impact populations, or as indirect effects wherein community structure may be altered because of toxicopathic losses of individuals. These latter impacts have been observed in benthic infaunal micro-invertebrates in Puget Sound (Long et al. 2005) but have been difficult to observe in higher organisms. Toxicopathic community effects in higher organisms such as fishes, birds and mammals are often modeled as cascade effects in the ecosystem based on known predator-prey or competitive relationships among affected species.

Persistent Bioaccumulative Toxics (PBTs)

Because many chemicals are persistent, bioaccumulative toxics (PBTs) understanding their fate and transport in the environment including movement in the food web is of paramount interest in evaluating threats. As reviewed in Chapter 2a, mammalian apex predators such as killer whales (Orcinus orca) and harbor seals (Phoca vitulina) have exhibited body burdens of persistent toxics (PCBs and PBDEs) expected to cause serious health effects (Ross 2006). Ross et al. (2000) characterized the Southern Resident Killer Whale population as among the most contaminated cetaceans in the world. Exposure to PBTs have been implicated as a cause for population decline in this population, as well as an impediment to their recovery (Krahn et al. 2002). PBT exposure in apex predators like these is widely thought to occur from consuming contaminated prey (Cullon et al. 2005, Cullon et al. 2009, O'Neill et al. 2006). The most highly PCB-contaminated populations of killer whale and harbor seal prey -- chinook salmon (O'Neill and West 2009) and Pacific herring (West et al. 2008) -- have been reported from Central and Southern Puget Sound.

Metals/organometals - Placeholder

Organochlorine pesticides - Placeholder

Other (Non-OC) Pesticides/Herbicides - Placeholder

Fossil fuels/PAHs - Placeholder

Dioxins/furans - Placeholder

Toxicopathic Impacts: three cases studies

The DPSIR conceptual model implies a left-to-right progression of thought and discovery from drivers to impacts. This type of model has directed a great deal of monitoring and assessment efforts to date in Puget Sound, including the PBT studies in fish and mammals described above. In some cases however, toxicopathic impacts have been identified in biota first, without knowledge of or understanding the drivers or pressures or conveyance pathways. In such cases, scientists have worked right-to-left from impacts to identify causative chemicals, pathways and sources. This approach requires field-biological capacity that can "…pay attention to unusual biological observations..", recognizing "…what is normal and abnormal…" (sensu, Sumpter and Johnson 2005) within the context of the range of stressors (pollution or other) that might cause such abnormalities. Three prominent indicators of biota health in Puget Sound that were developed in this manner are reviewed here as case studies.

Case 1: Cancerous liver tumors were observed in English sole (Pleuronectes vetulus), a bottom-dwelling flatfish, in Puget Sound's most polluted waters as early as 1975. At that time the disease was hypothetically linked to pollutant exposure. This cancerous biomarker has been used since that time as an indicator of bottomfish health in Puget Sound, and its cause has been identified as exposure to fossil fuels or by-products of their use (polycyclic aromatic hydrocarbons or PAHs -- Myers et al. 2003). Liver disease in English sole is being used to track efficacy of a sediment-PAH cleanup program in Eagle Harbor (Myers et al. 2008), and is currently being monitored along with sediment PAHs in Puget Sound to evaluate trends in ecosystem health Sound-wide. The disease is significant to fish because it is associated with reproductive impairment and liver disease that have fitness consequences (Johnson & Landahl 1993). One study suggested the level of impairment exhibited by English sole could reduce population size in exposed populations in Puget Sound (Johnson et al. 1998).

Case 2: Threats to bottomfish populations related to exposure to endocrine disrupting compounds (EDCs) have been identified in Puget Sound. Initially recognized in routine field monitoring efforts as abnormal gonadal development, specific toxicopathic reproductive anomalies such as abnormal spawn timing in male and females and feminization of male fish were later identified in English sole from Elliott Bay (Johnson et al. 2008). These authors noted that several EDC compounds that could cause these conditions have been identified in Elliott Bay sediments (Partridge et al., 2005), in watershed bodies, stormwater, and wastewaters draining to Elliott Bay. These compounds include both natural human estrogen (17-β estradiol) and synthetic estrogen (ethinylestradiol), which can be conveyed to aquatic systems via wastewater treatment plants in Puget Sound (Lubliner 2010), as well as nonylphenol (a surfactant commonly found in detergent) and bisphenol A (commonly found in polycarbonate plastics), which have been measured in stormwaters draining to Puget Sound (King County 2007).

Case 3: Contaminant threats to coho salmon (Oncorhynchus kisutch) spawning in urbanized, lowland stream reaches have been described from years of observing "pre-spawning mortality" of this species (McCarthy et al. 2008), wherein adults returning to spawn in such streams die before they can spawn, sometimes within a few hours of entering the stream. This threat is of particular concern because it affects a sensitive life history phase during reproduction, as coho salmon are moving from saltwater back to freshwater to spawn. This syndrome is associated with storm-related flash-flow regimes in lowland urban streams that receive stormwater draining from urban landscapes. Stormwater-conveyed contaminants and sedimentation have been implicated as causative, especially stormwater that occurs after a long antecedent dry spell.

State and Impact to Humans - Placeholder

Uncertainties and Information Gaps

The threat of toxics is related not only to the source, fate and transport of toxics in the environment, but also to the toxicity and subsequent harm to organisms. Significant uncertainties and knowledge gaps exist in all of these areas. Washington State agencies are currently placing a high emphasis on quantifying the type, loading amounts, and timing of toxic contaminants entering Puget Sound, especially via stormwater, and modeling the movement of toxics in the ecosystem. These ongoing efforts produce valuable estimates of contaminant loadings and information on how contaminants reach Puget Sound. In addition, this effort will produce associated estimates of uncertainty, which should be carefully considered in management responses.

Significant uncertainty also relates to gaps in knowledge, including:

- where and when accumulative toxics enter the food chain,
- temporal and spatial trends in biota-exposure for many contaminants, and
- the relative harm to biota and humans caused by exposures.

As described in Chapter 2 Biophysical status of Puget Sound, some of the greatest uncertainty regarding the threat of toxic chemical contaminants in the Puget Sound ecosystem is how toxics affect or harm organisms. Although there exists a great deal of information related to the extent and magnitude of exposure of Puget Sound biota to toxic contaminants, significant gaps in our understanding of how toxics harm biota include:

- toxicity of multiple-chemical mixtures,
- sublethal effects on reproduction and fitness,
- population-level effects,
- community-level effects related to changes in fitness and cascading competition and predation effects among affected species,
- realistic effects-thresholds for most Chemicals of Concern,
- the relative degree of threat for the wide range of toxics we are aware of, and

- exposure and effects in sensitive life-stages (such as eggs, larvae, and reproducing adults).

Careful selection of indicator species and metrics that can be used to evaluate these gaps will allow better understanding of where to focus limited recovery resources, as well as predict outcomes from recovery strategies.

Pressure: Toxic Chemical Contaminants in Freshwaters - Placeholder

- Ecology PBDE study
- Ecology Mercury/human health study
- Ecology PCB study?
- King Co. DNR Lake Washington EDC study
- NOAA salmon studies
- Copper
    - Current use pesticides

Driver: Intentional and Unintentional Introduction of Invasive and Non-native Species

Non-native species are those that do not naturally occur in an ecosystem. A non-native species is considered invasive when it is capable of aggressively establishing itself and causing environmental damage to an ecosystem. Plants, animals, and pathogens all can be invasive. Typical traits of an invasive species include: 1) generalist; being able to survive in a variety of physical and biological situations, 2) rapid reproduction, growth, and dispersal ability, and 3) lacking natural predators or pests in the invaded ecosystem. Thus, invasive non-native species are successful competitors in new ecosystems, usually displacing native species and disrupting ecosystem processes. An increase in invasive non-native species is associated with land cover change (human development and seral stage) and habitat fragmentation, human activities that transport the plants and animals or their eggs/seeds, and to changes in disturbance regimes (Hobbs 2000).

Invasive non-native species are a worldwide problem; in the United States alone an estimated 50,000 non-native species have either been introduced or escaped within natural or managed ecosystems (Pimentel et al 2000). With that many species involved, the fraction that is invasive does not have to be large to inflict great harm upon native species and natural ecosystems. For example, 602 of the 1055 native plant species and 68 out of 98 native bird species that are categorized as threatened in the United States are imperiled by invasive non-native species (Gurevitch and Padilla 2004).

Invasive non-native species are either introduced intentionally, with the express purpose being the translocation of the organism or unintentionally as a secondary byproduct (Ruiz and Carlton 2003). A few examples of unintentional introduction include: ballast water exchange, packing material, and pathogens hitchhiking on other organisms. Identifying pathways and vectors is critical because the easiest means to prevent and reduce the spread of new invasions is vector interception or disruption (Carlton and Ruiz 2005). Without managing the pathways and vectors by which invasive non-native species enter the Salish Sea ecosystem, the number of successful establishments of invasive non-native species will increase.

Placeholder: Economic Consequences

Pressure: Invasive and Non-native Species in Salish Sea Ecosystem

In the following section, we use DPSIR terminology to help evaluate invasive species as a pressure to the ecosystem in terms of impacts to native populations and communities with intentional and unintentional introductions as drivers (Figure 5). The strategies to control and prevent invasive species are discussed in more detail in Chapter 4.

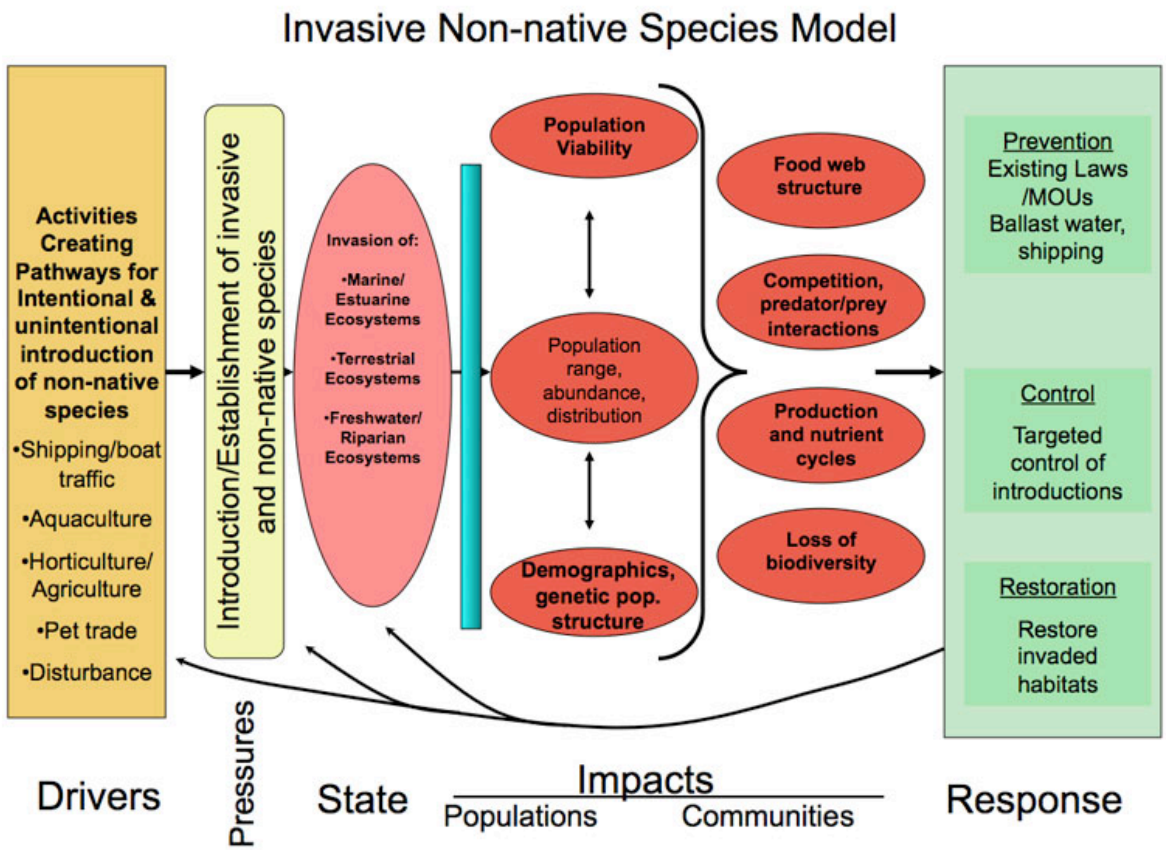

Figure 6. Driver-Pressure-State-Impacts-Response conceptual model for Invasive Species.

The Washington Invasive Species Council has identified approximately 700 invasive non-native species that have been introduced/established in and around Washington State at varying spatial extents (Washington Invasive Species Council 2009). Of these 700 species, the council identified 50 priority species/guilds based on these having highest impacts to the system. Of the top 50 priority species/guilds, 37 already occur in Washington and consist of 17 terrestrial plants, 2 terrestrial animals, 8 aquatic plants, 7 aquatic animals, and 3 insects/diseases. To give an idea as to the breadth of species considered in the top 50 invasive non-native species, a few examples include the following: knotweeds, butterfly bush, feral swine, spartina, caulerpa seaweed, New Zealand mud snail, tunicates, nutria, wood boring beetles, and viral hemorrhagic septicemia virus.

State: Invasion of Terrestrial Ecosystems

Clear differences have been demonstrated between invasive and non-invasive plant species on the basis of physiology, leaf-area allocation, shoot allocation, growth rate, size and fitness (van Kleunen et al 2010). Currently, the 2010 Washington State Noxious Weed list identifies 112 invasive non-native terrestrial plant species occurring throughout Washington . These species are classified into three major classes:

1. Class A- Composed of 28 invasive non-native species (5 are considered in the 50 priority species/guilds) with limited distribution in Washington and eradication is required by law.
2. Class B- Composed of 55 invasive non-native species (11 are considered in the 50 top priority species/guilds), which are presently established in limited portions of the state, with containment as the primary goal.
3. Class C- Composed of 29 invasive non-native species (1 is considered in the 50 top priority species/guilds) that are widespread in Washington, with flexibility of control at the local level.

For terrestrial animals there is no comparable comprehensive list of species present throughout the Puget Sound region like there is for aquatic environments. However, some individual counties have partial lists of non-prioritized invasive non-native species that are present. For example, King County identifies the European Starling, House Sparrow, Eastern Gray Squirrel and domestic cat as invasive within the county .

Impacts:

Two of the most influential factors influencing land invasions are disturbance and the transport of species by global trade. The facilitation between land transformation and global transport of species is bi-directionally linked. Land transformation provides opportunities for invasion and invasions can enhance and drive land transformation (Hobbs 2000). When both of these pressures interact, they create the potential for extreme changes of ecosystem dynamics (Hobbs 2000).

Fragmented native vegetation adjacent to human development is more likely to be invaded because of its interface with anthropogenic vegetation (edges) that enhances the spread of invasive species from the disturbed edges. The interaction between fragmentation and invasion results in changes in ecological processes, loss of native species and an overall reduction in biological diversity (Murcia 1995; With 2002). There is a coupling between disturbance (both natural and anthropogenic) and high levels of invasive non-native species. For example, Seattle public lands are highly disturbed by urbanization and also contain a large proportion of invasives. From 1999 to 2000, a citywide survey of Seattle's 3,215 hectares of public land was completed. The survey found that invasive non-native plant species are present in 94% of these urban natural areas and that 20% of the city's forested areas are highly invaded (Ramsay et al 2004). A follow-up sample performed in 2005 within the city's eight forest types, indicated that these lands were still highly invaded (Seattle Urban Nature Project 2006). In the mixed coniferous/deciduous forest type for example, the regenerating tree layer is composed of 55% non-native evergreen trees and 8% non-native deciduous trees. Overall biodiversity and ecosystem function typically become reduced when invasive species become dominant (Sanders et al 2003). With invasive non-native species composing more than half of all regenerating trees found, the forest is more susceptible to greater damage from disease, pests and other disturbances thereby jeopardizing the future of this forest type (Seattle Urban Nature Project 2005).

Farther from urban centers, clear-cut logging is a major source of disturbance in the Cascades (Parks et al 2005). Throughout the Coast Range and Western Cascades, invasives dominate for the first 2-5 years following the disturbance but are then replaced by native species as succession progresses. This pattern was observed from 2001-2005 across the Forest Inventory and Analysis Program plots in Washington and Oregon mountain ranges (Harrington et al 2007). They found that the percentage of invasive species declined with increasing stand size class. Since larger stand class size is highly associated with time since a major disturbance, most invasive plant species on forest land in the region are associated with recently-disturbed locations.

However, even though native plant species regain dominance over invasives after a forest disturbance, there are long term effects on the dominant native species (Dale and Adams 2003). During the first 10 years following the debris avalanche of Mount St. Helens, plots inundated with invasive species had significantly greater conifer sapling mortality and lower native species diversity than un-invaded plots. After this initial 10 year period, no difference in conifer mortality was noted and native species diversity was higher within invaded plots. Even so, the plots dominated by invasive species still had fewer conifer trees overall. Thus, the short-term conifer mortality increase associated with non-native species invasion appears to have long-term effects on the recovery of conifers as the dominant vegetation. The reduction of these foundational conifer species may cause cascading effects, affecting energy and nutrient fluxes, hydrology, biodiversity and food webs (Ellison et al 2005).

Global travel and commerce has redistributed species around the globe, connecting regions that historically were biogeographic barriers. A nation's non-native species diversity is strongly related to its level of trade (Westphal et al. 2008) and the United States is one of the leading nations in recipients of non-native invasives through international trade (Jenkins and Mooney 2006). While air- and seaports are major entry points for international trade, the commodities arriving at these destinations and those arriving from interstate commerce, are subsequently moved by road and rail. Many of these shipments contain unintended stowaways, such as untreated wood harboring non-native invasive insects and pathogens (Piel et al 2008). The travel corridors then help direct the movement of non-native invasives through less hospitable habitat, facilitating their spread and establishment (Hulme 2009).

Urban areas in the Puget Sound region are at high risk for introduction of non-native invasive bark- and wood-infesting insects based on the amount of urban and exurban forestland and the tonnage of imported goods (Colunga-Garcia et al 2009). The most prevalent pathway is imported machinery and nonmetallic mineral products originating from Asia (Colunga-Garcia et al 2009). There have been several incidences of Asian and citrus long-horned beetles being found in warehouse and plant nursery shipments to Washington State. Some beetles escaped into neighboring greenbelts, necessitating the cutting of several thousand trees, injections of a systemic pesticide, and the quarantine of all host material for a one-half mile radius around the beetle introduction site4.

Other non-native invasives are spread intentionally for human use, for example agriculture, horticulture or pet trade. The majority of woody non-native invasive plants in the United States were introduced for horticultural purposes—82% of 235 woody plant species identified as colonizing outside of cultivation have been used in landscaping (Reichard and White 2001). A conservative number of 104 non-native invasive shrub species are known in the United States, with at least 17 species occurring in Washington State (Boyce 2009). Many of these shrubs affect native forests by crowding out native species, reducing biodiversity and may change ecosystem functioning effectively halting successful tree regeneration (Boyce 2009). Shrubs are often introduced by escaping from gardens, where they are grown for their flowers and fruit. Birds and mammals are responsible for furthering the spread of the shrubs due to the large fruit crops produced. Birds alone are vectors for the seed dispersal of over 70 non-native shrub species nationwide (Boyce 2009).

State: Invasion of Marine/Estuarine Ecosystems

Coastal estuarine and marine ecosystems are among the most heavily invaded systems in the world (Grosholz 2002), mostly due to intentional and unintentional introductions by boat traffic (ballast water and hulls), aquaculture, bait and released pets (Carlton 2000).

Even though estuarine and marine systems are heavily invaded, currently, the 2010 Washington State Noxious Weed list identifies only 4 invasive non-native estuarine/marine plant species occurring throughout Puget Sound . However, all 4 are considered Class A species meaning eradication is required by law.

The Washington Department of Fish and Wildlife maintains a watch list of aquatic nuisance species of Washington's marine and freshwaters. These non-native species are considered to have a high risk of becoming invasive and are separated into species of primary concern—those considered to have the highest level of environmental risk, and secondary concern—considered to have a lower level of environmental risk. According to the 2008 watch list, of the marine animals, 8 of 9 primary and 22 of 34 secondary species of concern are currently present in Washington. All but two of the primary marine species of concern overlaps with the top 50 priority species/guilds listed by the Washington Invasive Species Council. While a total of 4 marine secondary species of concern overlaps with the top 50 priority species.

Impacts:

A study by Lawrence and Cordell (2010) looked at how ballast water influences the amount of propagules (e.g., larvae) of non-native species found in Puget Sound waters. Cordell's results indicate that the Puget Sound receives an annual average of 7.5 x 106 m3 of ballast water from both foreign (mostly trans-pacific) and domestic waters. Foreign trans-Pacific vessels carried significantly fewer ($p < 0.001$) propagules compared to ships on domestic west coast routes. Of the propagules detected, trans-Pacific ships contained almost twice as many non-native species (19 species) than did those from ships on west coast routes (10 species), with seven species being common to both. However, even though trans-Pacific vessels had higher diversity of non-native species, densities of nonnatives were 100-200% greater in domestic ballast water. Considering that a variety of biological and physical factors affect an invader's success, both foreign (high diversity) and domestic (high density) sources of ballast water have high potential to result in successful invasions of the Sound.

Wonham and Carlton (2005) reviewed the literature documenting 123 introduced invertebrate, algal, fish and vascular plant species in the Northeastern Pacific Ocean. They found the major invasion pathways to be shipping (hull fouling, solid and water ballast) and shellfish (particularly oysters) and finfish imports. Successful invasions increased at linear, quadratic, and expotential rates for different taxa, pathways and regions in the Northeastern Pacific. Of the regions included in this study, Puget Sound had the most introduced species.

Ballast water is not the only vector for distributing invasive non-native species in the Sound. Of the 62 established invertebrate invasive non-native taxa found in Puget Sound waters, only 25 are spread by ballast (Simkanin et al 2009). Six of these 25 taxa are exclusively distributed by ballast. Other major sources include ship fouling (35 taxa) and commercial oysters (39 taxa) (Simkanin et al 2009). Sixteen and 17 taxa are distributed exclusively by fouling and commercial oysters respectively.

Regardless of invasion pathway, invasive non-native marine/estuarine species in the Puget Sound are capable of causing extensive ecological changes. For example, highly invasive non-native cordgrass (S. alterniflora, S. anglica, S. patens, and S. densiflora) in estuarine habitat rapidly converts bare mudflat into a cordgrass monoculture. S. alterniflora was accidentally introduced in the 1890s when it was used as packing material for oysters shipped from the Atlantic coast (Grevstad et al 2003), it is most widely spread in Willapa Bay, infesting approximately 465 solid hectares (Phillips et al 2008). S. anglica was introduced in Port Susan Bay in 1961 for erosion control and cattle forage and infested approximately 36 solid hectares (Phillips et al 2008). S. patens and S. densiflora introduction pathways are unknown and take up less than 0.40 solid hectare at the mouth of the Dosewallips river and Gray's Harbor (Phillips et al 2008). These infestations of Spartina have negative community level effects as it greatly reduces habitat available for fish, shellfish (commercial and native), migratory waterfowl and shorebirds (Hacker et al 2001; Buchanan 2003; Grevstad et al 2003; Semmens 2008).

When Spartina invades a variety of potential niches, physical conditions of the habitat are the main limiting factors controlling the high variation in establishment and growth among habitats rather than biological interactions (Dethier and Hacker 2005). Thus, the range, abundance and physical and biological effects of Spartina do vary depending on the type of habitat invaded (Hacker and Dethier 2006). Of four habitat types considered (mudflat, cobble beach, low and high salinity marsh), Spartina has the greatest range and abundance in mudflats and low salinity marshes compared to high salinity marshes and cobble beaches. Changes in sediment characteristics also substantially differed among habitats; some habitats experience greater accretion (mudflats), greater water content (cobble beach), and greater salinity loss (high salinity) than other habitats. Finally, native plant diversity declined in low salinity marshes but either increased or remained stable within the other habitat types, although percent cover and species richness of native macroalgae decreased. Thus, if changes occur in salinity, sea level, or sediment supply in various invaded habitats, Spartina impacts will be altered, most likely to the detriment of the native community (Hacker and Dethier 2006).

Another estuarine invasive non-native ecological engineer, an eelgrass (Z. japonica), has had an opposite community effect. Eelgrass beds provide habitat and food to a wide variety of marine organisms, protection for fry, and prevent beach erosion, thus being a critical component of the nearshore ecosystem. The invasive form of eelgrass typically does not coexist or compete with the native eelgrass (Z. marina) of Puget Sound but simply extends the eelgrass bed further into the upper intertidal zone (Britton-Simmons et al 2010). Within two decades of introduction, Z. japonica almost doubled the total eelgrass habitat in Boundary Bay, British Columbia (Williams

2007). Now migrating waterfowl prefer it over native eelgrass as their principal food. Z. japonica has increased faunal diversity, net primary production and influenced the biogeochemistry of the entire estuary (Williams 2007).

These two invasive non-native species, S. alterniflora and Z. japonica also have indirect effects on each other (Williams 2007). The vector of the interaction depends on which colonizes first. If S. alterniflora colonizes first, it outcompetes Z. japonica. However, if Z. japonica colonizes first it inhibits the seed germination of S. alterniflora.

Oysters are another ecosystem engineer, having major impacts on coastal ecosystems (Ruesink et al 2005). Not only are the oysters food for fish and invertebrates, they also improve water quality by filtration and provide habitat by creating biogenic reefs. These reefs influence water flow, which alters sediment deposition, consolidation, and stabilization. Thus, oysters can have disproportionately high impacts on the ecosystem, although impacts vary by species. Two important species will be considered here:

1. The Olympia oyster (Ostrea lurida); native to Puget Sound but became commercially unviable due to overharvesting in the late 1800s, and despite a century of negligible harvesting, it remains commercially unviable to this day (Trimble et al 2009).
2. The non-native Pacific oyster (Crassostrea gigas); commercially replaced O. lurida in 1928 and is now Washington's most valuable shellfish resource (Dethier 2006).

The lack of recovery by native O. lurida is partially due to competition with C. gigas and other non-native and invasive species (Trimble et al 2009). Interspecific competition reduced Olympia oyster survival with C. gigas growing at twice the rate of native O. lurida (Buhle and Ruesink 2009; Trimble et al 2009). Fouling organisms, most of them non-native, kill or reduce food access to O lurida. The removal of fouling organisms, doubles the chance that O. lurida will survive and improves its growth (Trimble et al 2009). One particular invasive non-native of note that affects both O. lurida and C. gigas is the oyster drill (Ocinebrina inornata). This species was introduced before 1965 with shipments of Pacific oysters, is now established and widespread in Willapa Bay and is a significant pest of oyster aquaculture (Buhle and Ruesink 2009). Where drills are present they reduce overall survival of both oyster species, killing on average 0.33 (SE = 0.08) Pacific oysters, and 0.16 (SE = 0.04) Olympia oysters per drill per week, dependent upon prey density.

Placeholder: Green crabs, tunicates

State: Invasion of Freshwater/Riparian Ecosystems

Currently, the 2010 Washington State Noxious Weed list identifies 26 invasive non-native freshwater plant species occurring throughout Washington. These species are classified into three major classes:

1. Class A- Composed of 7 invasive non-native species (2 are considered in the 50 priority species/guilds) with limited distribution in Washington and eradication is required by law.

2. Class B- Composed of 14 invasive non-native species (4 are considered in the 50 top priority species/guilds), which are presently established in limited portions of the state, with containment as the primary goal.
3. Class C- Composed of 5 invasive non-native species (None are considered in the 50 top priority species/guilds) that are widespread in Washington, with flexibility of control at the local level.

The Washington Department of Fish and Wildlife maintains a watch list of aquatic nuisance species of Washington's marine and freshwaters. These non-native species are considered to have a high risk of becoming invasive and are separated into species of primary concern—those considered to have the highest level of environmental risk, and secondary concern—considered to have a lower level of environmental risk. According to the 2008 watch list, of the freshwater animals, 3 of 14 primary and 12 of 39 secondary species of concern are currently present in Washington. All but two of the primary freshwater species of concern overlaps with the top 50 priority species/guilds listed by the Washington Invasive Species Council. However, only one of the freshwater secondary species of concern overlaps with the top 50 priority species.

Impacts:

Riparian zones are significant because of their ameliorating influence on aquatic ecosystems. These zones are unique ecological hotspots; instrumental in providing shelter and food for aquatic organisms, stream temperature regulation, maintaining healthy water quality by filtering contaminants and stabilizing the soil (Gregory et al 1991; Naiman 2005). When invasive non-native species displace natives within the riparian zone, these introduced species have the potential to cause long-term cascading changes in the structure and functioning of both the riparian zone and adjacent aquatic habitat. A study by Urgenson et al (2009) found the invasive non-native giant knotweed (Polygonum sachalinense) caused such changes in community function and structure of western Washington riparian zones. Richness and abundance of native herbs, shrubs, and juvenile trees were negatively correlated with knotweed density, with a 70% reduction of native leaf litter mass. Knotweed litter has a carbon:nitrogen ratio of 52:1, which is 38-58% higher than that of native woody species. Knotweed invasion, with its litter of lower nutritional quality could affect the productivity of macro-invertebrate communities and in turn, the fish and other animals that use these invertebrates as a primary food source. Other effects of knotweed, such as decline in regeneration of red alder (a nitrogen fixer) and conifers, have important implications for nitrogen cycling and amount of large woody debris respectively.

Washington State lake ecosystems have an invasion history involving the introduction and establishment of numerous plants and animals. One species of special importance within the Puget Sound Basin is crayfish. Crayfish are a keystone species capable of effecting changes in primary productivity, food web dynamics, water quality, and biodiversity (Mueller 2007). Washington State has one native species, the signal crayfish (Pacifastacus leniusculus). However, two invasive species in Washington have been documented; in the year 2000, an invasive species, the red swamp crayfish (Procambarus clarkia) was discovered (Mueller 2002) and 2007 marked the first sighting of the northern crayfish (Orconectes virilis) (Larson and Olden 2008). As of 2008, of 58 lakes surveyed in the Puget Sound region, P. clarkia was found in ten and O. virilis was found in three (Larson and Olden 2008). The lakes that are invaded by P. clarkia, are clustered near schools which use crayfish in their science programs whereas the lakes invaded by O. virilis were all near golf courses, with ponds at golf courses oftentimes being stocked with crayfish for aquatic macrophyte control (Larson and Olden 2008). The close association between the schools and golf courses with the invaded lakes provides strong support for implicating them as introduction pathways for crayfish.

Another introduced species within the lake system is the Chinese mystery snail (Bellamya chinensis). These snails were introduced 40 years ago and are now broadly distributed in hundreds of lakes that historically supported relatively few native snails (Olden et al 2009). The Chinese mystery snail has now become a food source for both the native and invasive species of crayfish. Interestingly, the native crayfish is able to consistently handle and consume the snails at a faster pace, outcompeting both species of invasive crayfish for this novel food source (Olden et al 2009). Even so, the invasive red swamp crayfish still outnumbers the native signal crayfish by more than 2 to 1 where both species co-occur (Mueller 2007). Thus, the likelihood that red swamp crayfish will alter freshwater aquatic ecosystems in the Pacific Northwest is high (Moore 2006).

Freshwater fish species are introduced around the world due to demands for aquaculture (39%), ornamental fish (21%), modification of wild stocks (17%), sport fishing (12%), accidentally (8%) and biocontrol/engineering (6%) (Gozlan et al 2010). In western North America in particular, a variety of non-native fishes and the bullfrog (Rana catesbeiana) have been widely introduced, mainly for aquaculture and sport fishing (Adams 1999). One major impact from these introductions is the loss or decline of native amphibian species. Amphibian species richness is significantly lower at ponds having non-native predatory fish present than at either non-predatory or fish-free ponds (Hecnar and M'Closkey 2001). Even in otherwise relatively pristine high mountain lakes, lower amphibian diversity occurs in lakes harboring introduced trout, with long-toed salamanders and pacific treefrogs most negatively impacted (Bull 2002).

Non-native fish species facilitate the viability of another non-native freshwater predator, the bullfrog (citation). The bullfrog is a formidable predator, with large specimens capable of preying upon small birds, young snakes, crayfish, other frogs, and minnows. Non-native fish prey upon native macroinvertebrates thus indirectly facilitating the survival of bullfrog tadpoles (Adams et al 2003). Further, in pond surveys the best predictors of bullfrog abundance were the presence of non-native fish. However, when comparing the effects of non-native fishes and bullfrogs on red-legged frogs in the lowlands of Western Washington, red-legged frogs were significantly impacted more by non-native fishes than bullfrogs (Adams 1999). Thus, in the Pacific Northwest, non-native fish may pose a greater conservation concern than bullfrogs, at least for amphibians (Richter and Azous 1995; Adams 1999; Adams et al 2003).

Place holder: Millfoil is another important invasive in freshwater system. Himalayan blackberry, Scotch broom

1. Summary, Uncertainties and Information Gaps

Approximately 700 invasive species occur near or in the Puget Sound/Georgia Basin, many of which have become established in our native ecosystems (Washington Invasive Species Council

2009). With so many species involved, it becomes necessary to prioritize control efforts based upon ecological and economic impact. However, prioritization is no easy task considering multiple taxa and habitat types are involved and interaction/facilitation between species occurs. Some comparative studies have been attempted, but are far from being comprehensive (Adams 1999). Ranking systems appear to be useful for more comprehensive prioritization (Randall et al. 2008), although they are based on expert opinion and current knowledge. This is the method used by NatureServe (2004) and multiple states, including Washington. The assessment tool used by the Washington Invasive Species Council allows invasive non-native species to be ranked according to their ecological impact and the likelihood of Washington state agencies being able to effectively implement prevention measures or conduct early action on a species.

Ecosystem Models

Many types and classes of models have been developed and applied to parts or all of the Salish Sea ecosystem including efforts to model impacts of climate change (e.g., Kairis 2010, Casola 2009), assess the implications of alternative urban growth patterns (e.g., Alberti et al. 2004), predict impacts of future seismic events (e.g. Hyndman 2003, Hartzell 2002), predict weather patterns (e.g., Grell et al. 1995, Colle 1998), understand water circulation patterns (e.g., Hamilton 1985, Babson et al. 2006), evaluate residency time of toxic chemicals and effects on biota (e.g. Spromberg 2006), assess food web dynamics (e.g., Harvey et al. 2010), predict biological invasions (e.g., Cordell 2010, Colnar 2007), etc. Because of immediate information needs, we focus our efforts on the models that identify and compare threats to the Salish Sea ecosystem and identify indicators of threats or ecosystem condition in this Science Update. Secondarily and incompletely, we focus on models that help identify mechanistic linkages between threats (Drivers) and changes in ecosystem states but only for the "high" and "very high" ranked threats that we identified in the Introduction of this chapter. Finally, we present water circulation models because they are focused on identifying the cause of an event (low oxygen levels in Hood Canal) of concern to many because of the negative effects on biota. This event may be associated with one or more of the high level threats but until the cause is better understood, we will not know.

For our purposes, a "model" is a mathematical representation of the ecosystem or components of the ecosystem including human impacts. For the models described we identify model inputs, their primary findings, their utility to management and conservation, and their ability to identify ecosystem threats and indicators. In addition, when information is available, we provide information on model reliability, which is usually assessed by comparing simulated results to empirical data or correlations between derived indices and biological data. Finally, we identify information gaps.

1. Models Identifying Ecosystem Threats and Indicators

For the ecosystem threats and indicator models that follow we compare their potential to be used to identify /evaluate threats in Table 5 below. We also assess their ability to identify indicators or for the model outputs to be used as indicators.

Relative Risk Models

Relative risk models were developed to characterize relative risks to an ecosystem and have been used for a variety of purposes (see Landis and Wiegers 2007). These models have been applied to very large estuaries to evaluate the relative influence of different stressors (threats) and their sources (e.g., Iannuzzi et al. 2009). In Puget Sound, this modeling approach has been used to investigate the causes of the Cherry Point Pacific Herring run (Landis et al. 2004) and to identify, rank and assess their combined impacts of stressors to the near shore environment at Cherry Point (Hayes and Landis 2004). The Cherry Point near shore analysis, analyzed cumulative impacts from multiple sources of chemical and non-chemical stressors (e.g., ballast water, piers, point source pollution, recreational activities) to assess risk to multiple species that use the near shore environment (Hayes and Landis 2004). This approach allows researchers to compare threats spatially and quantitatively and to identify: (1) the most threatened geographical subregions, (2) the sources contributing the most risk, and (3) the habitats and speices most at risk. To date, this model has only been applied at small scales but could be applied to the entire Salish Sea ecosystem. Results from this modeling effort suggest that the major contributors of risk to the Cherry Point near shore marine environment are vessel traffic, upland urban and agricultural land use, and shoreline recreational activities (Hayes and Landis 2004). For the Cherry Point Pacific herring stock, exploitation, habitat alteration and climate change were the risk factors that contributed to the decline. The retrospective assessment identified the warm Pacific Decadal Oscillation (PDO) as the primary factor altering herring population dynamics (Landis et al. 2004).

Mass-balance Model for evaluating Food Web Structure and Community Scale Indicators

Harvey et al. (2010) developed a mass-balance model of the Puget Sound Central Basin food web with the goal of identifying indicators for assessing the effectiveness of various management activities. The model consists of 65 functional groups that range from primary producers to top order consumers that live in nearshore, offshore, pelagic, and demersal environments. It also includes several fisheries. Their model indicates that the system is dominated by demersal species and that most of the biomass is aggregated in seven functional groups. Bottom-up dynamics appear to strongly influence trophic flows but there are examples of top-down control with bald eagles apparently able to cause trophic cascades. Model simulations indicate that current commercial fishing mortality appears to be sustainable and below maximum sustained yields due, in part, to declines in commercial fisheries in recent years. Their model has not yet been used to test the ecosystem-level impacts of past levels of fishing effort on previously heavily exploited fishes such as rockfish and gadoids. Finally, their model has significant implications on which species or functional groups are good indicators of changes in management activities (e.g., Samhouri et al. 2009) but their technical memo does not contain recommended indicators. Other than fisheries, this model does not include the impacts of human activities on the food web and is currently focused on central Puget Sound but will include other basins in future iterations and will eventually be replaced by an Atlantis model (Horne et al. 2010, Fulton et al. 2004, 2007). The Atlantis model will add several features that the current model is lacking, most notably: tighter coupling between functional groups and abiotic features like temperature, circulation, nutrients and dissolved oxygen; spatial dynamics that allow simulation of multiple basins of Puget Sound; species-habitat interactions; and more realistic representation of life history features such as age structure, migrations, and prey switching. Atlantis also enables simulation of monitoring and assessment programs designed to evaluate the effectiveness of management policies.

Mapping Cumulative Impacts to the California Current Marine Ecosystems

Halpern et al. (2008) developed an ecosystem-specific, multiscale spatial model in a GIS environment that combined multiple drivers (e.g., sea temperature, shipping, and species invasion) into a single estimate of cumulative human impact for the world's oceans. In a second paper, Halpern et al. (2009) focused on mapping these same cumulative impacts to California Current marine ecosystems with the goal of identifying the most and least impacted areas and the top threats to the region - this analysis included Puget Sound. However, results for Puget Sound proper were not discussed.

The highest impact scores were concentrated around areas of large human populations including Puget Sound. Climate change drivers (SST, UV, and ocean acidification) exhibited the greatest ecosystem impacts across the region because of their widespread distribution and high vulnerability of many ecosystems to these stressors. Other important drivers included atmospheric deposition of pollution, ocean-based pollution, and commercial shipping. Intertidal and nearshore ecosystems were the most heavily impacted because of exposure to stressors from both land- and ocean-based human activities. The two top impacted ecosystems by human activities were mudflats and oyster reefs. The authors attribute the impacts to these systems from historic overharvesting of oysters and subsequent disease outbreaks that accompanied the introduction of non-native and invasive species and to the expansion of non-native species like cordgrass (Spartina alterniflora) into mudflats (Callaway and Josselyn 1992, Ruesink et al. 2005). Other highly impacted ecosystems identified by Halpern et al. (2009) included salt marsh, beach, seagrass, and rocky intertidal.

Mapping the Terrestrial Anthropogenic Impacts to the Western U.S. – Human Footprint

In the terrestrial enviornament, researchers have evaluated the cumulative impact of human activities in a GIS environment at global (e.g., Sanderson et al. 2002), national (Theobald 2010) and regional (e.g., Leu et al. 2008) scales. The regional effort by Leu et al. (2008), involved calculating the physical human footprint, defined as the actual space occupied by human features for the western U.S. including Washington. Recognizing that human features influence ecological processes beyond the physical space occupied by those features they also mapped the effect area, or the ecological human footprint. To accomplish this, they derived an index that combines 14 landscape structural and anthropogenic features in a GIS environment: human habitation, interstate highways, federal and state highways, secondary roads, railroads, irrigation canals, power lines, linear feature densities, agricultural lands, campgrounds, highway rest stops, landfills, oil and gas developments, and human-induced fires.

They estimated that 13% of the western U.S. was dominated by human features with agricultural land, human population areas, and roads covering the majority of this area. In addition, they found that low elevation areas with deeper soils were disproportionally affected (43% vs. 7%) by the human footprint and so were ecoregions dominated by urbanized areas like the Puget Trough - Willamette Valley - Georgia Basin ecoregion.

To test the footprint model, they correlated bird abundance patterns with human footprint patterns and found that synanthropic species increased with greater human footprint scores and species sensitive to habitat fragmentation generally decreased in abundance with increasing human footprint scores (Leu et al. 2008, Johnson et al. 2010, Knick and Hanser 2010). In addition, the presence of a deadly fungal disease (Batrachochytrium dendrobatidis) in native frogs of the Pacific Northwest was strongly correlated with human footprint scores (Adams et al. 2010) – this disease has been associated with rapid global decline and extinction of amphibians in several regions around the world (Skerratt et al. 2007). Like the California Current model above, the human footprint model can be used to identify areas for conservation activities, areas for restoration and areas appropriate for human activity. In addition, the authors of this model developed a theoretical approach to using human footprint data to monitor the effectiveness of landscape level conservation efforts (Haines et al. 2008).

Models Associated with the Threat Climate Change

Many models have been developed to assess climate change impacts on plants and animals, hydrology, sea surface temperature, weather patterns, sea level, ocean acidification, UV radiation, etc. In addition there have been several efforts to summarize and synthesize the findings of these models (e.g., Climate Impacts Group 2009, and IPCC 2007). The Climate Change section of this chapter focuses on the outcomes of climate change models and rather than repeat this information here, we refer readers to that section or to the reports that summarize and explain the various modeling efforts.

Models Associated with the threat Residential, Commercial and Industrial Development

The Distributed Hydrology Soil Vegetation Model (DHSVM) is a spatially explicit, biophysically-driven hydrologic model (Wigmosta et al. 1994, 2002; Cuo et al. 2008, 2009). DHSVM uses GIS-derived representations of elevation, soil type, soil thickness, vegetation, and meteorological data to simulate water and energy fluxes at and below the land surface. The model has been used to evaluate effects of forest management on land surface hydrologic response, especially flooding, of upland forested basins (e.g., Bowling and Lettenmaier 2001; Lamarche and Lettenmaier 2001; Whitaker et al. 2003). The model represents the effects of topography on incident and reflected solar radiation, and on downslope redistribution of moisture in the saturated zone, which in turn controls both fast and slow runoff response. DHSVM has been recently modified to predict the hydrologic response of partially urbanized watersheds by altering the treatment of precipitation on impervious surfaces, adding water detention, and spatially varying the surface runoff depending on land cover (Cuo et al. 2008, 2009). The model's output, which closely matches empirically observed trends in flux rates and volumes, illustrates important linkages between landscape pattern and hydrology, with more extreme, episodic flux rates and volumes in urbanizing, highly impervious landscapes (Bowling and Lettenmaier 2001; Lamarche and Lettenmaier 2001; Whitaker et al. 2003; Cuo et al. 2008, 2009).

The Land Cover Change Model (LCCM) was developed to forecast potential trends in land cover change in the central Puget Sound, in conjunction with landscape-based models of bird species richness and abundance (Hepinstall et al. 2008, 2009). LCCM uses a set of spatially explicit multinomial logit models of site-based land-cover transitions. LCCM is fully integrated with the UrbanSim model (Waddell et al. 2003), a spatially explicit socioeconomic model of land use decision-making that predicts changes in the spatial distribution of households, jobs, and real estate quantities, types, and prices. Coupling the LCCM with UrbanSim allows for simulation of multiple interacting aspects of urban development, via UrbanSim's interfaces with external macroeconomic and transportation models. The LCCM explicitly models human decisions responsible for land cover change including interactions among humans and between socioeconomic and environmental variables, and dynamic shifts in land use/land cover resulting from such interactions (Hepinstall et al. 2008, 2009). Model results suggest that, under current development trends, urban land cover is expected to increase over the next 20 years by 68-73 percent of its 1999 extent, with resultant shifts (based on linkages to avian diversity models) in the dominance of synanthropic and early successional guilds over forest interior bird guilds (Hepinstall et al. 2008, 2009).

Envisioning Puget Sound Alternative Futures (Bolte and Vache). See Models Associated with the Threat Shoreline Development below.

Models Associated with the Threat Shoreline Development

Historic change and impairment of Puget Sound shorelines (Simenstad et al. 2009) – this change analysis modeled changes in the spatial arrangement of dominant ecosystem processes along Puget Sound's beaches, estuaries and river deltas. The outcomes of this change analysis are described in detail under Shoreline Development.

Envisioning Puget Sound Alternative Futures (Bolte and Vache 2010). This effort models changes in landscape composition based on alternative trajectories: 1) status quo: continuation of current trends, 2) Managed growth: concentrates growth within urban growth areas and near regional growth centers, and 3) unmanged growth: relaxation of land use restrictions. Scenarios were created using a spatially and temporally explicit alternative futures model and created a set of spatial coverages reflecting different scenario outcomes for a variety of landscape variables (land use/land cover, shoreline modifications, and population projections). The model also generated a set of summary statistics describing landscape change variables. This modeling effort is being used to project future impairment of ecosystem functions, goods and services. Results are presented by sub-basins. The results are presented in 12 maps and 57 graphs that generally demonstrate greater loss of forests, wetlands and an increase in development associated with the unmanged growth scenario but with considerable variation among sub-basins. In addition, graphs indicate an increase in docks, impervious survaces, marinas, and shoreline armoring associated with unmanged growth.

Models Associated with the Threat Pollution

Placeholder – this section needs to be developed.

Models Associated with the Threat Invasive and Non-native Species

Physico-chemical factors affecting copepod occurrences

Cordell et al. (2010) modeled the physio-chemical factors affecting occurrences of non-indigenous planktonic copepod in the northeast Pacific estuaries. They characterized estuaries with and without populations of the copepod Pseudodiaptomus inopinus and identified relatively low salinity and stratification of water column temperature and salinity as important predictors of copepod occurrence. This type of modeling can be used to predict species invasions and environmental susceptibility and potentially identify methods to reduce invasion potential.

Please see Invasive and Non-native Species section for other modeling efforts.

Puget Sound Water Circulation and Water Quality Models

In some cases we don't know the threats but observe events that cause concern and then attempt to identify the cause (usually viewed as a threat). The following modeling efforts attempt to identify the causes of ecological events such as low oxygen events in Hood Canal that cause negative effects on the biota. Modeling efforts are particularly useful for these types of investigations because of complex interactions among a variety of factors contributing to the event including bathymetry, water circulation, and water chemistry. We recommend expanding this section to include additional models especially by those involved in these efforts.

Hood Canal Dissolved Oxygen Program

The deep waters of the southern Hood Canal have historically had low dissolved oxygen concentration. However, in recent years the severity of the hypoxia has increased and is having negative effects on biota. In response, the Hood Canal Dissolved Oxygen Program was initiated to investigate the sources of low oxygen events and their effects on marine life. Researchers are using the Regional Ocean Modeling System of Haidvogel et al. (2008) to achieve this goal and publications are expected in the next year and should be available for future editions of this publication.

Estuarine circulation model for Puget Sound, Georgia Basin

Researchers with the Puget Sound Regional Synthesis Model initiative (PRISM 2010) developed an estuarine circulation model for Puget Sound, Georgia Basin and linkages to Pacific currents, adapted from the Princeton Ocean Model (POM; Edwards et al. 2007). The model is designed to examine hydrodynamic factors including three-dimensional patterns of water column circulation, tidal and riparian fluxes, water temperature, and salt water/freshwater exchange patterns and rates within the Puget Sound/Georgia Basin system. Simulation results have been favorably compared with empirical measurements for Carr Inlet (Edwards et al. 2007), and are being used to demonstrate that surface current patterns and other hydrodynamic factors potentially play a significant role in driving hypoxic conditions in Hood Canal (Hood Canal Dissolved Oxygen Program, unpublished results). The model is being used more broadly by PRISM to understand temporal dynamics in salt/freshwater exchanges, differences in subbasin residence times, and contributions of freshwater fluxes from Puget Sound/Georgia Basin to oceanic currents.

'''Summary and Conclusions

Although incomplete, we found that ecosystem modeling efforts are being broadly applied to the Salish Sea ecosystem to help us understand everything from the relative magnitude of ecosystem threats to the causes of low oxygen events. In this Update we identify models that identify and rank threats to the Salish Sea ecosystem and that can be used as indicators or can be used to identify a potential suite of indicators and provide a summary of those efforts in Table 5.

During this model identification and review process, we identify the following research needs:

1. In Table 5, we assess the use of various models to identify and rank threats and identify indicators. The approaches described here or similar approaches could be applied at the scale of the Salish Sea ecosystem. Such an effort would identify the primary threats, help quantify the extent of ecosystem threats, and identify the most threatened ecosystems. This type of information is critical for spatially explicit and effect conservation planning. Ideally, the terrestrial and aquatic modeling efforts would be combined into a single seamless model of the marine, terrestrial and freshwater ecosystems because of the interacting and synergistic effects of threats originating and moving between these ecosystems.
2. Expand the mass balance model to the entire Salish Sea and eventually replace it with the Atlantis model. This effort will allow managers to identify effective indicators at the scale of the Salish Sea and the use of the Atlantis model will allow better coupling between functional groups and abiotic features like temperature, circulation, nutrients and dissolved oxygen; spatial dynamics that allow simulation of multiple basins of Puget Sound; species-habitat interactions; and more realistic representation of life history features such as age structure, migrations, and prey switching. Atlantis also enables simulation of monitoring and assessment programs designed to evaluate the effectiveness of management policies
3. Continue to link modeling efforts as demonstrated by the linking of cycling and circulation models to investigate causes of low oxygen events. Such links allow researchers to expand the scope and scale of inference and take advantage of existing efforts.
4. When causes of ecosystem change are not well understood, as is the case with low oxygen levels in Hood Canal, models can be used to understand the causes of these types of events.

Table 5. Models identifying ecosystem threats and indicators we assess their intended or potential use to identify/evaluate threats or management alternatives and their ability to identify or be used as indicators.

Model	Objectives	Inputs	Used to identify/evaluate threats?	Used to identify indicators?	Used as an indicator?
Relative Risk Models	Characterize risks to ecosystems from various stressors including threats to fish runs and impacts from chemical stressors	Categorical ranks for each stressor (spills, land use, ballast water, etc.). Inputs include volume, percent cover, number of various stressors. Habitat types are also included in the model using length and area	Yes – quantitatively identifies threats to ecosystems	Potentially	Potentially – could use change in stressor ranks

		by type. Risk predictions are point estimates based on ranks and effects of parameter uncertainty is assessed using a Monte Carlo Analysis			
Mass balance food web	Identify indicators for assessing the effectiveness of various management activities	65 functional groups that range from primary producers to top order consumers that live in nearshore, offshore, pelagic, and demersal environments. It also includes several fisheries.	Yes -fisheries only	Yes - the primary objective	Potentially
California Current	Identifying the most and least impacted areas and the top threats to the California Current region	Combined 25 anthropogenic drivers (e.g., sea temperature, shipping, pollution, fisheries and species invasion) into a single estimate of cumulative human impact	Yes - quantitatively identifies primary anthropogenic threats to the marine ecosystem	Yes	Yes
Human footprint	Map the extent of anthropogenic features and their extended area of influence for the western U.S. To assess the human footprint extent	Index that combines 14 landscape structural and anthropogenic features in a GIS environment: human	Yes - quantitatively identifies the combined anthropogenic impacts to terrestrial and freshwater	Yes	Yes - theoretical approach published

among ecoregions, freshwater aquatic systems, across lands differing in ownership and protection status and across physical environmental gradients (e.g., productivity and elevation). Goal was to identify areas where management actions could reduce human influences, to locate areas for restoration, to evaluate “what if” scenarios, and to assess changes in the human footprint over time	habitation, interstate highways, federal and state highways, secondary roads, railroads, irrigation canals, power lines, linear feature densities, agricultural lands, campgrounds, highway rest stops, landfills, oil and gas developments, and human-induced fires	systems

Conclusion

1. What are the biggest threats to the health of Puget Sound?

We reviewed eight assessments of threats relevant to the Salish Sea ecosystem. While each presents a unique list, there is considerable overlap and consistent high ranking of development, climate change, invasive species, pollution, and shoreline modification. Species harvesting was also highly ranked and should be priority topics for future synthesis.

In just over a century, the human enterprise in the Salish Sea Ecosystem has had tremendous impacts. The human footprint has taken roughly half of the forest and wetlands, impounded 37% of the drainage, removed 1000 km of natural shoreline and altered weather patterns such that entire glaciers have been lost. Simultaneously, human activities have introduced toxins, endocrine disruptors, and at least 700 non-native species in the system. The combined impacts of these changes and their current and future interactions in an environment substantially warmed by anthropogenic energy demands are profound and wide reaching.

Conversion of land from forest to human settlements has transformed the watersheds that feed the Salish Sea to the detriment of terrestrial, freshwater, and marine ecosystems. The importance of linkages between terrestrial and aquatic systems cannot be overstated. Furthermore, the interaction between factors such as modifications to the landscape and climate change can enhance declines of habitats (e.g., salmon habitat).

What are key lessons learned?

In an effort to boil down the information in this chapter and highlight the most important lessons learned, we selected key pieces of information where there was a significant weight of evidence to support the observation, there were good data and high certainty, impacts were wide ranging impacts in the ecosystem or were derived from multiple threats, and where good information existed to characterize threats (please see specific sections for references).

Climate change If climate changes as predicted, the following impacts will likely occur:

- The Climate Impacts Group predict average temperature rise in the Pacific Northwest of 1.1°C.
- The combination of warming temperatures and decreased snow to precipitation ratio will affect snowmelt and the region's water supply will be affected.
- Stream temperature will be less hospitable to salmon.
- Sea surface temperatures in Puget Sound will increase by ~ 6°C, causing an increase in algal blooms and hypoxic events.
- Acidification of Puget Sound waters is cause for concern for organisms at the base of the food chain supporting higher tropic levels.

Invasive species

- Of the 700 species introduced/established in and around Washington State, the council identified 50 priority species/guilds based on these having highest impacts to the system.
- Trans-Pacific vessels had higher diversity of non-native species, and densities of non-natives were 100-200% greater in domestic ballast water. Considering that a variety of biological and physical factors affect an invader's success, both foreign (high diversity) and domestic (high density) sources of ballast water have high potential to result in successful invasions of the Sound.
- In the Pacific Northwest, non-native fish may pose a greater conservation concern than bullfrogs, at least for amphibians.
- Identifying pathways and vectors is critical because the easiest means to prevent and reduce the spread of new invasions is vector interception or disruption.
- For terrestrial animals there is no comparable comprehensive list of species present throughout the Puget Sound region like there is for aquatic environments.

Residential, Commercial and Industrial Development

- The biophysical contrasts introduced through the process of residential, commercial and industrial development – particularly through the replacement of native vegetation with impervious surfaces – impact ecological processes from the ecosystem to species level.
- Development within the Puget Sound watershed, particularly within the central Sound region, has increased at a rate of approximately 1.4 percent per year over the last decade, and is forecasted to spread well into the Cascade foothills by 2027.
- Canges in land cover and land use result in significant loss of nutrient and water retention, affecting water quality and quantity in the Salish Sea ecosystem. Simultaneously, such changes introduce new stressors through introduction of chemical contaminants and increased stormwater runoff.

Shoreline development

- Approximately 99.8 percent of shoreline exhibit some level modification and degradation to nearshore processes.
- Of the forms of shoreline modification, shoreline armoring is most prevalent, comprising 74 percent of all artificial shoreforms.
- Shoreline modification has resulted in significant disruption or loss of important natural shoreforms such as large river deltas, coastal embayments, beaches and bluffs, and estuarine wetlands. Shorelines have also exhibited considerable shortening and simplification as a consequence of modification.
- Dominant impacts of shoreline modification include disruption of sedimentation rates and patterns, which affect the geomorphology and maintenance of shoreform structures.
- Ecosystemic changes resulting from shoreline modification lead to significant disruption or loss of plant and animal habitats, particularly affecting salmonids and other important aquatic species.

Pollution

- Non-point source pollution carried by stormwater and atmospheric processes represent the greatest threat of contaminant loadings from their terrestrial sources to Puget Sound.
- Residential pollution sources are a large contributor to toxics in Puget Sound.
- The probability of a catastrophic oil spill in Puget Sound is low but the threat of long-term damage from such an event is high.
- A wide range of Chemicals of Concern for Puget Sound has been identified based on a broad range of conveyance pathways and contaminant types, and on the threat of harm to biota health.
- Trophic transfer (food-web biomagnification) of persistent bioaccumulative toxics has resulted in high threat of toxicopathic disease to apex predators such as southern resident killer whales and harbor seals in Puget Sound.
- Toxicopathic cancer in English sole from habitats along urbanized shorelines is caused by exposure to fossil-fuel compounds in their environment, and is being used to track bottom fish health through time.
- English sole from habitats throughout Puget Sound have shown reproductive impairment related to exposure to endocrine disrupting compounds, possibly related to human wastewater.
- Pre-spawing mortality of coho salmon returning to some urbanized stream is linked to stormwater runoff from urban landscapes.

Future Directions We view this threat assessment as a "work in progress" and hope that other contributors will fill in missing pieces (e.g., threats to human wellbeing, and other identified threats not covered) using peer-reviewed sources, provide additional information and help identify any mistakes or misrepresentation of information.

Our review of the literature suggests that threats have been identified and classified along broad categories (Table 1). The scientific community did a good job at developing conceptual models using DPSIR framework to show linkages among threats. We tried to build on these efforts by providing information to validate those links. However, we identify the need to further demonstrate linkages and the effect of interactions among threats in a more quantitative way. We propose methods to accomplish this effort in the Introduction and Ecosytem Models sub-sections.

We also identify the need for a more comprehensive, quantitative and systematic assessment of the relative magnitude of threats and the uncertainty surrounding the relative magnitude of threats. We did not find a peer-reviewed analysis of the relative magnitude of threats for Puget Sound proper. Therefore, our Chapter treats threats separately and does evaluate the relative importance of threats on the Puget Sound ecosystem. However, we identified modeling approaches that help identify and compare threats quantitatively and the information contained in this chapter will hopefully contribute the information needed to build such models. Ideally, the models would help tease apart the confounding effects of human activities and natural events. The output of the modeling exercise would be to provide recommendation on priorities for management and policy.

More effort is needed to translate threats into measures or indicators of threats following Haines et al. 2008.

Ideally, future threats assessments would be both spatially and temporally explicit. For example, GIS maps of contamination would be comprehensive and demonstrate levels of contamination explicitly highlighting when contamination levels exceed heath thresholds or impair population survival and reproduction.

References

Adam, J. C., A. F. Hamlet, and D. P. Lettenmaier. 2009. Implications of global climate change for snowmelt hydrology in the twenty-first century. Hydrological Processes 23:962-972.

Adams, M. J. 1999. Correlated Factors in Amphibian Decline: Exotic Species and Habitat Change in Western Washington. The Journal of Wildlife Management 63:1162-1171.

Adams, M. J., N. D. Chelgren, D. Reinitz, R. A. Cole, L. J. Rachowicz, S. Galvan, B. McCreary, C. A. Pearl, L. L. Bailey, J. Bettaso, E. L. Bull, and M. Leu. 2010. Using occupancy models to understand the distribution of an amphibian pathogen, Batrachochytrium dendrobatidis. Ecological Applications 20:289-302.

Adams, M. J., C. A. Pearl, and R. Bruce Bury. 2003. Indirect facilitation of an anuran invasion by non-native fishes. Ecology Letters 6:343-351.

Agee, J. K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC.

Ahmad, Z., J. R. Herman, A. P. Vasilkov, M. Tzortziou, B. G. Mitchell, and M. Kahru. 2003. Seasonal variation of UV radiation in the ocean under clear and cloudy conditions [5156-10]. PROCEEDINGS- SPIE THE INTERNATIONAL SOCIETY FOR OPTICAL ENGINEERING:63-73.

Alberti, M. 2005. The effects of urban patterns on ecosystem function. International Regional Science Review 28:168-192.

Alberti, M. 2008. Advances in Urban Ecology: Integrating Humans and Ecological Processes in Urban Ecosystems. Springer, New York, NY.

Alberti, M. 2009. Central Puget Sound 2007 Land Cover Classification. NSF Biocomplexity II Grant 2005-2009. Urban Landscape Patterns: Complex Dynamics and Emergent Properties. Puget Sound Regional Synthesis Model (PRISM). University of Washington.

Alberti, M., D. Booth, K. Hill, B. Coburn, C. Avolio, S. Coe, and D. Spirandelli. 2007. The impact of urban patterns on aquatic ecosystems: An empirical analysis in Puget lowland sub-basins. Landscape and Urban Planning 80:345-361.

Alberti, M. and J. M. Marzluff. 2004. Ecological resilience in urban ecosystems: Linking urban patterns to human and ecological functions. Urban Ecosystems 7:241-265.

Alberti, M., J. M. Marzluff, E. Shulenberger, G. Bradley, C. Ryan, and C. Zumbrunnen. 2003. Integrating humans into ecology: opportunities and challenges for studying urban ecosystems. Bioscience 53:1169-1179.

Alberti, M., R. Weeks, and S. Coe. 2004. Urban land-cover change analysis in Central Puget Sound. Photogrammetric Engineering and Remote Sensing 70:1043-1052.

Alig, R. J., J. D. Kline, and M. Lichtenstein. 2004. Urbanization on the US landscape: looking ahead in the 21st century. Landscape and Urban Planning 69:219-234.

Anderson, E. M., Lovvorn, J.R., D. Esler, W. S. Boyd, and K. C. Stick. 2009. Using predator distributions, diet, and condition to evaluate seasonal foraging sites: sea ducks and herring spawn. Marine Ecology Progress Series 386:287-302.

Andersson, A. J., F. T. Mackenzie, and N. R. Bates. 2008. Life on the margin: Implications of ocean acidification on Mg-calcite, high latitude and cold-water marine calcifiers. Marine ecology progress series. 373:265.

Antonov, J. I., S. Levitus, and T. P. Boyer. 2005. Thermosteric sea level rise, 1955-2003. Geophys. Res. Lett. 32:L12602.

Babson, A. L., A. Kawase, and P. MacCready. 2006. Seasonal and interannual variability in the circulation of Puget Sound, Washington: A box model study. Atmosphere-Ocean 44:29-45.

Baker, L. A., D. Hope, Y. Xu, J. Edmonds, and L. Lauver. 2001. Nitrogen balance for the central Arizona-Phoenix (CAP) ecosystem. Ecosystems 4:582-602.

Baker, M. E., M. J. Wiley, and P. W. Seelbach. 2001. GIS-based hydrologic modeling of riparian areas: Implications for stream water quality. Journal of the American Water Resources Association 37:1615-1628.

Banas, N. S., E. Lessard, R. Kudela, P. MacCready, T. Peterson, B. M. Hickey, and E. Frame. 2009. Planktonic growth and grazing in the Columbia River plume region: A biophysical model study. Journal of Geophysical Research 114:C00B06.

Barrett, N. E. and W. A. Niering. 1993. Tidal marsh restoration: trends in vegetation change using a geographical information system (GIS). Restoration Ecology 1:18-28.

Beamer, E. M., A. McBride, R. Henderson, and K. Wolf. 2003. The importance of non-natal pocket estuaries in Skagit Bay to wild Chinook salmon: an emerging priority for restoration. Skagit River System Cooperative, LaConner, WA.

Beechie, T., E. Buhle, M. Ruckelshaus, A. Fullerton, and L. Holsinger. 2006. Hydrologic regime and the conservation of salmon life history diversity. Biological Conservation 130:560-572.

Beechie, T., B. D. Collins, and G. Pess. 2001. Holocene and recent changes to fish habitats in two Puget Sound basins. Pages 37-54 in J. M. Dorava, B. Palcsak, F. Fitzpatrick, and D. R. Montgomery, editors. Geomorphic Processes and Riverine Habitat. American Geophysical Union, Washington, DC.

Bessie, W. C. and E. A. Johnson. 1995. The relative importance of fuels and weather on fire behavior in subalpine forests. ECOLOGY -NEW YORK- 76:747.

Bilby, R. E. and L. A. Mollot. 2008. Effect of changing land use patterns on the distribution of coho salmon (Oncorhynchus kisutch) in the Puget Sound region. Canadian Journal of Fisheries and Aquatic Sciences 65:2138-2148.

Bilkovic, D. M. and M. M. Roggero. 2008. Effects of coastal development on nearshore estuarine nekton communities. Marine Ecology-Progress Series 358:27-39.

Blewett, C. M. and J. M. Marzluff. 2005. Effects of urban sprawl on snags and the abundance and productivity of cavity-nesting birds. Condor 107:678-693.

Bolte, J. and K. Vache. 2010. Envisioning Puget Sound Alternatives Futures. Prepared for Washington Department of Fish and Wildlife in support of the Puget Sound Nearshore Ecosystem Restoration Project. Prepared by Department of Biological and Ecological Engineering, Oregon State University, Corvallis, OR.

Booth, D. B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of the North American Benthological Society 24:724-737.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38:835-845.

Booth, D. B., J. R. Karr, S. Schauman, C. P. Konrad, S. A. Morley, M. G. Larson, and S. J. Burges. 2004. Reviving urban streams: Land use, hydrology, biology, and human behavior. Journal of the American Water Resources Association 40:1351-1364.

Bottom, D. L., K. K. Jones, T. J. Cornwell, A. Gray, and C. A. Simenstad. 2005. Patterns of Chinook salmon migration and residency in the Salmon River estuary (Oregon). Estuarine, Coastal, and Shelf Science 64:79-93.

Bowling, L. C. and D. P. Lettenmaier. 2001. The effects of forest roads and harvest on catchment hydrology in a mountainous maritime environment. Pages 145-164 in M. S. Wigmosta and S. J. Burges, editors. Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas. AGU Water Science and Application, Volume 2.

Boyce, R. 2009. Invasive Shrubs and Forest Tree Regeneration. Journal of Sustainable Forestry 28:1-2.

Brett, M. T., G. B. Arhonditsis, S. E. Mueller, D. M. Hartley, J. D. Frodge, and D. E. Funke. 2005. Non-point-source impacts on stream nutrient concentrations along a forest to urban gradient. Environmental Management 35:330-342.

Britton-Simmons, K. H., S. Wyllie-Echeverria, E. K. Day, K. P. Booth, K. Cartwright, S. Flores, C. C. Garcia, T. L. Higgins, C. Montanez, and A. Rames. 2010. Distribution and Performance of the Nonnative Seagrass Zostera japonica across a Tidal Height Gradient on Shaw Island, Washington. PACIFIC SCIENCE 64:187-198.

Brockmeyer, J., R.E., J. R. Rey, R. W. Virnstein, R. G. Gilmore, and L. Earnest. 1997. Rehabilitation of impounded estuarine wetlands by hydrologic reconnection to the Indian River Lagoon, Florida (USA). Wetlands Ecology and Management 4:93-109.

Brown, L. R., T. F. Cuffney, J. F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A. H. Bell, and J. T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051-1069.

Bryant, J. C. and R. H. Chabreck. 1998. Effects of impoundment on vertical accretion of coastal marsh. Estuaries 21:416-422.

Buchanan, J. B. 2003. Spartina invasion of Pacific coast estuaries in the United States: implications for shorebird conservation. BULLETIN- WADER STUDY GROUP 100:47-49.

Buchanan, J. B. 2006. Nearshore Birds in Puget Sound. Report No. 2006-05, Puget Sound Nearshore Partnership, Seattle, WA.

Buhle, E. R. and R. J.L. 2009. Impacts of invasive oyster drills on olympia oyster (Ostrea lurida carpenter 1864) recovery in Willapa Bay, Washington, United States. J. Shellfish Res. Journal of Shellfish Research 28:87-96.

Bull, E. L. and D. B. Marx. 2002. Influence of Fish and Habitat on Amphibian Communities in High Elevation Lakes in Northeastern Oregon. Northwest science : official publication of the Northwest Scientific Association. 76:240-248.

Callaway, J. C. and M. N. Josselyn. 1992. The introduction and spread of smooth cordgrass (Spartina alterniflora) in south San Francisco Bay. Estuaries 15:218-226.

Camfield, A. F., S. F. Pearson, and K. Martin. 2010. Life history variation between high and low elevation subspecies of horned larks Eremophila spp. Journal of Avian Biology 41:273-281

Canning, D. J. and H. Shipman. 1995. Coastal erosion management studies in Puget Sound, Washington: Executive Summary. Vol. 1. Report #94-74, Water and Shorelands Program, Washington Department of Ecology, Olympia, WA.

Carlton, J. T. 2000. Global Change and Biological Invasions in the Oceans.in H. A. Mooney and R. J. Hobbs, editors. Invasive species in a changing world. Island Press, Washington, D.C.

Carlton, J. T. and G. M. Ruiz. 2005. Vector science and integrated vector management in bioinvasion ecology: conceptual frameworks.in H. A. Mooney and I. C. f. S. S. C. o. P. o. t. Environment., editors. Invasive alien species : a new synthesis. Island Press, Washington, DC.

Carr, E. R., P. M. Wingard, S. C. Yorty, M. C. Thompson, N. K. Jensen, and J. Roberson. 2007. Applying DPSIR to sustainable development. International Journal of Sustainable Development and World Ecology 14:543-555.

Carter, T., C. R. Jackson, A. Rosemond, C. Pringle, D. Radcliffe, W. Tollner, J. Maerz, D. Leigh, and A. Trice. 2009. Beyond the urban gradient: barriers and opportunities for timely studies of urbanization effects on aquatic ecosystems. Journal of the North American Benthological Society 28:1038-1050.

Case, M. and D. Peterson. 2005. Fine-scale variability in growth climate relationships of Douglas-fir, North Cascade Range, Washington. Canadian Journal of Forest Research 35:2743-2755.

Casola, J. H., L. Cuo, B. Livneh, D. P. Lettenmaier, M. T. Stoelinga, P. W. Mote, and J. M. Wallace. 2009. Assessing the Impacts of Global Warming on Snowpack in the Washington Cascades. JOURNAL OF CLIMATE 22:2758-2772.

CBP. 2010. Bay Pressures, http://www.chesapeakebay.net/baypressures.aspx?menuitem=13959. Chesapeake Bay Program, Annapolis, MD.

Chappell, C. B., M. S. M. Gee, B. Stephens, R. Crawford, and S. Farone. 2001. Distribution and decline of native grasslands and oak woodlands in the Puget lowland and Willamette Valley ecoregions, Washington. Pages 124-139 in S. H. Reichard, P. W. Dunwiddie, J. G. Gamon, A. R. Kruckeberg, and D. L. Salstrom, editors. Conservation of Washington's Native Plants and Ecosystems. Washington Native Plant Society, Seattle, WA.

Church, J., N. White, T. Aarup, W. Wilson, P. Woodworth, C. Domingues, J. Hunter, and K. Lambeck. 2008. Understanding global sea levels: past, present and future. Sustainability Science 3:9-22.

Churkina, G., D. G. Brown, and G. Keoleian. 2010. Carbon stored in human settlements: the conterminous United States. Global Change Biology 16:135-143.

Clampitt, C. A. 1993. Effects of human disturbances on prairies and the regional endemic Aster curtus in western Washington. Northwest Science 67:163-169.

Clancy, M., I. Logan, J. Lowe, J. Johannessen, A. MacLennan, F. B. Van Cleve, J. Dillon, B. Lyons, R. Carman, P. Cereghino, B. Barnard, C. Tanner, D. Myers, R. Clark, J. White, C. A. Simenstad, M. Gilmer, and N. Chin. 2009. Management Measures for Protecting the Puget Sound Nearshore. Report No. 2009-01, Puget Sound Nearshore Ecosystem Restoration Project, Olympia, Washington.

Cleaves, D. A. 1994. Assessing uncertainty in expert judgments about natural resources. Report No. GTR SO-110, U.S. Department of Agriculture, Forest Service, Southern Forest Experiment Station.

Climate Impacts Group. 2009. The Washington climate change impacts assessment evaluating Washington's future in a changing climate. Climate Impacts Group, [Seattle, Wash.].

Colle, B. A. and C. F. Mass. 1998. Windstorms along the western side of the Washington Cascade Mountains. Part I: A high-resolution observational and modeling study of the 12 February 1995 event. Monthly Weather Review 126:28-52.

Colle, B. A., C. F. Mass, and B. F. Shull. 1999. An observational and numerical study of a cold front interacting with the Olympic Mountains during COAST IOP5. Monthly Weather Review 127:1310-1334.

Collins, B. D. and D. R. Montgomery. 2002. Forest development, wood jams and restoration of floodplain rivers in the Puget Lowland. Restoration Ecology 10:237-247.

Collins, B. D., D. R. Montgomery, and A. D. Haas. 2002. Historical changes in the distribution and functions of large wood in Puget Lowland rivers. Canadian Journal of Fisheries & Aquatic Sciences 59:66-76.

Colnar, A. M. and W. G. Landis. 2007. Conceptual model development for invasive species and a regional risk assessment case study: The European green crab, Carcinus maenas, at Cherry Point, Washington, USA. Human and Ecological Risk Assessment 13:120-155.

Colunga-Garcia, M., R. A. Haack, and A. O. Adelaja. 2009. FOREST ENTOMOLOGY - Freight Transportation and the Potential for Invasions of Exotic Insects in Urban and Periurban Forests of the United States. Journal of economic entomology. 102:237.

Cooke, R. M. and L. H. J. Goossens. 2004. Expert judgement elicitation for risk assessments of critical infrastructures. Journal of Risk Research 7:643-656.

Cordell, J. R., L. M. Tear, and S. M. Bollens. 2010. Modelling physico-chemical factors affecting occurrences of a non-indigenous planktonic copepod in northeast Pacific estuaries. Biological Invasions 12:1427-1445.

Crawford, R. C. and H. Hall. 1997. Changes in the south Puget Sound prairie landscape. Pages 11-15 in P. Dunn and K. Ewing, editors. Ecology and Conservation of the South Puget Sound Prairie Landscape. The Nature Conservancy of Washington, Seattle, WA.

Crooks, K. R. 2002. Relative sensitivities of mammalian carnivores to habitat fragmentation. Conservation Biology 16:488-502.

Crooks, K. R. and M. E. Soule. 1999. Mesopredator release and avifaunal extinctions in a fragmented system. Nature 400:563-566.

Cullon, D. L., S. J. Jeffries, and P. S. Ross. 2005. Persistent organic pollutants in the diet of harbor seals (Phoca vitulina) inhabiting Puget Sound, Washington (USA), and the Strait of

Georgia, British Columbia (Canada): a food basket approach. Environmental Toxicology and Chemistry 24:2562-2572.

Cullon, D. L., M. B. Yunker, C. Alleyne, N. J. Dangerfield, S. O'Neill, M. J. Whiticar, and P. S. Ross. 2009. Persistent organic pollutants in Chinook salmon (Oncorhynchus tshawytscha): Implications for resident killer whales of British Columbia and adjacent waters. Environmental Toxicology and Chemistry 28:148-161.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Cuo, L., D. P. Lettenmaier, M. Alberti, and J. E. Richey. 2009. Effects of a century of land cover and climate change on the hydrology of the Puget Sound basin. Hydrological Processes 23:907-933.

Dale, V. H. and W. M. Adams. 2003. Plant reestablishment 15 years after the debris avalanche at Mount St. Helens, Washington. The Science of The Total Environment 313:101-113.

Davis, M. B. 1973. Pollen evidence of changing land use around the shores of Lake Washington. Northwest Science 47:133-148.

deMaynadier, P. G. and M. L. Hunter. 2000. Road effects on amphibian movements in a forested landscape. Natural Areas Journal 20:56-65.

Dennison, W. C. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27:15-26.

Derraik, J. G. B. 2002. The pollution of the marine environment by plastic debris: a review. Marine Pollution Bulletin 44:842-852.

Dethier, M. N. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Report No. 2006-04, Puget Sound Nearshore Partnership, Seattle, WA.

Dethier, M. N. and S. D. Hacker. 2005. Physical Factors vs. Biotic Resistance in Controlling the Invasion of an Estuarine Marsh Grass. Ecological Applications 15:1273-1283.

Dethier, M. N., United States. Army. Corps of Engineers. Seattle District., and Puget Sound Nearshore Partnership. 2006. Native shellfish in nearshore ecosystems of Puget Sound. Seattle District, U.S. Army Corps of Engineers, Seattle, Wash.

DoE, W. 2010. Overview: Threats – Saving Puget Sound, http://www.ecy.wa.gov/puget_sound/threats.html. Washington State Department of Ecology, Olympia, WA.

Domingues, C. M., J. A. Church, N. J. White, P. J. Gleckler, S. E. Wijffels, P. M. Barker, and J. R. Dunn. 2008. Improved estimates of upper-ocean warming and multi-decadal sea-level rise. 453:1090-1093.

Doney, S. C., V. J. Fabry, R. A. Feely, and J. A. Kleypas. 2009. Ocean Acidification: The Other CO2 Problem. doi:10.1146/annurev.marine.010908.163834. Annual Review of Marine Science 1:169-192.

Donnelly, R. and J. M. Marzluff. 2004. Importance of reserve size and landscape context to urban bird conservation. Conservation Biology 18:733-745.

Donnelly, R. and J. M. Marzluff. 2006. Relative importance of habitat quantity, structure, and spatial pattern to birds in urbanizing environments. Urban Ecosystems 9:99-117.

Doten, C. O., L. C. Bowling, J. S. Lanini, E. P. Maurer, and D. P. Lettenmaier. 2006. A spatially distributed model for the dynamic prediction of sediment erosion and transport in mountainous forested watersheds. Water Resources Research 42.

Downing, J. 1983. The Coast of Puget Sound: Its Processes and Development. University of Washington Press, Seattle, WA.

Dugan, J. E. and D. M. Hubbard. 2006. Ecological responses to coastal armoring on exposed sandy beaches. Shore and Beach 74:10-16.

Dunn, P. and K. Ewing. 1997. Ecology and Conservation of the South Puget Sound Prairie Landscape. The Nature Conservancy of Washington, Seattle, WA.

Dutch, M., V. Partridge, S. Weakland, K. Welch, and E. Long. 2009. Quality Assurance Project Plan: The Puget Sound Assessment and Monitoring Program Sediment Monitoring Component. Washington State Department of Ecology.

Edwards, K. A., M. Kawase, and C. P. Sarason. 2007. Circulation in Carr Inlet, Puget Sound, during spring 2003. Estuaries and Coasts 30:945-958.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin 44:III-VII.

Ellison, A. M., M. S. Bank, B. D. Clinton, E. A. Colburn, K. Elliott, C. R. Ford, D. R. Foster, B. D. Kloeppel, J. D. Knoepp, G. M. Lovett, J. Mohan, D. A. Orwig, N. L. Rodenhouse, W. V. Sobczak, K. A. Stinson, J. K. Stone, C. M. Swan, J. Thompson, B. Von Holle, and J. R. Webster. 2005. Loss of Foundation Species: Consequences for the Structure and Dynamics of Forested Ecosystems. Frontiers in Ecology and the Environment 3:479-486.

EnviroVision Corporation, Herrera Environmental Consultants Inc., and Washington Department of Ecology. 2008. Control of toxic chemicals in Puget Sound Phase 2: Pollutant loading estimates for surface runoff and roadways. Washington Department of Ecology.

EPA, U. 2010. Puget Sound Georgia Basin Ecosystem, http://www.epa.gov/Region10/psgb/. Region 10, U.S. Environmental Protection Agency, Seattle, WA.

Ewald, G. P., H. Larsson, L. Linge, and N. Szarzi. 1998. Biotransport of organic pollutants to an inland Alaska Lake by migrating sockeye salmon (Oncorhynchus nerka). Arctic 51:40-47.

Fabry, V. J., B. A. Seibel, R. A. Feely, and J. C. Orr. 2008. Impacts of ocean acidification on marine fauna and ecosystem processes. ICES Journal of Marine Science 65:414-432.

Faeth, S. H., P. S. Warren, E. Shochat, and W. A. Marussich. 2005. Trophic dynamics in urban communities. Bioscience 55:399-407.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. Journal of Wildlife Management 61:603-610.

Fahrig, L. and T. Rytwinski. 2009. Effects of Roads on Animal Abundance: an Empirical Review and Synthesis. Ecology and Society 14:20.

Feely, R. A., S. R. Alin, J. Newton, C. L. Sabine, M. Warner, A. Devol, C. Krembs, and C. Maloy. 2010. The combined effects of ocean acidification, mixing, and respiration on pH and carbonate saturation in an urbanized estuary. Estuarine, Coastal and Shelf Science 88:442-449.

Feely, R. A., C. L. Sabine, J. M. Hernandez-Ayon, D. Ianson, and B. Hales. 2008. Evidence for Upwelling of Corrosive "Acidified" Water onto the Continental Shelf. Science. 320:1490.

Feely, R. A., C. L. Sabine, K. Lee, W. Berelson, J. Kleypas, V. J. Fabry, and F. J. Millero. 2004. OCEAN SCIENCE - Impact of Anthropogenic CO2 on the CaCO3 System in the Oceans. Science. 305:362.

Feminella, J. W. and C. J. Walsh. 2005. Urbanization and stream ecology: an introduction to the series. Journal of the North American Benthological Society 24:585-587.

Field, C. B., M. L.D, M. Brklacich, D. L. Forbes, P. Kovacs, J. A. Patz, R. S.W, and M. J. Scott. 2007. North America.in M. L. Parry and Intergovernmental Panel on Climate Change. Working Group II., editors. Climate change 2007 : impacts, adaptation and vulnerability : contribution of Working Group II to the fourth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, U.K.; New York.

Finlayson, D. P., Washington Sea Grant Program., and Puget Sound Nearshore Partnership. 2006. The geomorphology of Puget Sound beaches. Washington Sea Grant Program, University of Washington, [Seattle, Wash.].

Frank, L. D., B. Stone, and W. Bachman. 2000. Linking land use with household vehicle emissions in the central puget sound: methodological framework and findings. Transportation Research Part D: Transport and Environment 5:173-196.

Fresh, K. L. 2006. Juvenile Pacific Salmon in Puget Sound. Report No. 2006-06, Puget Sound Nearshore Partnership, Seattle, Washington.

Fulton, E. A., J. S. Parslow, A. D. M. Smith, and C. R. Johnson. 2004. Biogeochemical marine ecosystem models II: the effect of physiological detail on model performance. Ecological Modelling 173:371-406.

Fulton, E. A., A. D. M. Smith, and C. R. Johnson. 2004. Biogeochemical marine ecosystem models I: IGBEM - a model of marine bay ecosystems. Ecological Modelling 174:267-307.

Fulton, E. A., A. D. M. Smith, and D. C. Smith. 2007. Alternative management strategies for southeast Australian Commonwealth fisheries: Stage 2: quantitative management strategy evaluation. CSIRO Australia, Tasmania.

Garthwaite, P. H., J. B. Kadane, and A. O'Hagan. 2005. Statistical methods for eliciting probability distributions. Journal of the American Statistical Association 100:680-700.

Gaydos, J. K., L. Dierauf, G. Kirby, D. Brosnan, K. Gilardi, and G. E. Davis. 2008. Top 10 Principles for Designing Healthy Coastal Ecosystems Like the Salish Sea. Ecohealth 5:460-471.

Gehrt, S. D. and S. Prange. 2007. Interference competition between coyotes and raccoons: a test of the mesopredator release hypothesis. Behavioral Ecology 18:204-214.

Giles, L. J. 1970. The ecology of the mounds on Mima Prairie with special reference to Douglas-fir invasion. M.S. Thesis. University of Washington, Seattle, WA.

Gozlan, R. E., J. R. Britton, I. Cowx, and G. H. Copp. 2010. Current knowledge on non-native freshwater fish introductions. Journal of Fish Biology 76:751-786.

Gray, D. 2008. The relationship between climate and outbreak characteristics of the spruce budworm in eastern Canada. Climatic Change 87:3-4.

Gregory, J. H., M. D. Dukes, P. H. Jones, and G. L. Miller. 2006. Effect of urban soil compaction on infiltration rate. Journal of Soil and Water Conservation 61:117-124.

Grell, G. A., J. Dudhia, and D. R. Stauffer. 1995. A Description of the Fifth-Generation Penn State/NCAR Mesoscale Model (MM5). NCAR Technical Note NCAR/TN-398 + STR, Mesoscale and Microscale Meteorology Division, National Center for Atmospheric Research, Boulder, CO.

Grevstad, F. S., D. R. Strong, D. Garcia-Rossi, R. W. Switzer, and M. S. Wecker. 2003. Biological control of Spartina alterniflora in Willapa Bay, Washington using the planthopper Prokelisia marginata: agent specificity and early results. Biological Control 27:32-42.

Griggs, G. B. 2005. The Impacts of Coastal Armoring. Shore and Beach 73:13-22.

Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global Change and the Ecology of Cities. Science 319:756-760.

Grimm, N. B., R. W. Sheibley, C. L. Crenshaw, C. N. Dahm, W. J. Roach, and L. H. Zeglin. 2005. N retention and transformation in urban streams. Journal of the North American Benthological Society 24:626-642.

Groffman, P. M., N. J. Boulware, W. C. Zipperer, R. V. Pouyat, L. E. Band, and M. F. Colosimo. 2002. Soil nitrogen cycle processes in urban Riparian zones. Environmental Science & Technology 36:4547-4552.

Groffman, P. M. and M. K. Crawford. 2003. Denitrification potential in urban riparian zones. Journal of Environmental Quality 32:1144-1149.

Groffman, P. M., A. M. Dorsey, and P. M. Mayer. 2005. N processing within geomorphic structures in urban streams. Journal of the North American Benthological Society 24:613-625.

Groffman, P. M., N. L. Law, K. T. Belt, L. E. Band, and G. T. Fisher. 2004. Nitrogen fluxes and retention in urban watershed ecosystems. Ecosystems 7:393-403.

Groffman, P. M., R. V. Pouyat, M. L. Cadenasso, W. C. Zipperer, K. Szlavecz, I. D. Yesilonis, L. E. Band, and G. S. Brush. 2006. Land use context and natural soil controls on plant community composition and soil nitrogen and carbon dynamics in urban and rural forests. Forest Ecology and Management 236:177-192.

Grosholz, E. 2002. Ecological and evolutionary consequences of coastal invasions. Trends in Ecology & Evolution 17:22-27.

Guinotte, J. M. and V. J. Fabry. 2008. Ocean Acidification and Its Potential Effects on Marine Ecosystems. Annals of the New York Academy of Sciences 1134:320-342.

Gurevitch, J. and D. K. Padilla. 2004. Are invasive species a major cause of extinctions? Trends in Ecology & Evolution 19:470-474.

Hacker, S. D. and M. N. Dethier. 2006. Community modification by a grass invader has differing impacts for marine habitats. Oikos 113:279-286.

Hacker, S. D., D. Heimer, C. E. Hellquist, T. G. Reeder, B. Reeves, T. J. Riordan, and M. N. Dethier. 2001. A Marine Plant (Spartina Anglica) Invades Widely Varying Habitats: Potential Mechanisms of Invasion and Control. Biological Invasions 3:211-217.

Haidvogel, D. B., H. Arango, W. P. Budgell, B. D. Cornuelle, E. Curchitser, E. Di Lorenzo, K. Fennel, W. R. Geyer, A. J. Hermann, L. Lanerolle, J. Levin, J. C. McWilliams, A. J. Miller, A. M. Moore, T. M. Powell, A. F. Shchepetkin, C. R. Sherwood, R. P. Signell, J. C. Warner, and J. Wilkin. 2008. Ocean forecasting in terrain-following coordinates: Formulation and skill assessment of the Regional Ocean Modeling System. Journal of Computational Physics 227:3595-3624.

Haines, A. M., M. Leu, L. K. Svancara, J. M. Scott, and K. P. Reese. 2008. A theoretical approach to using human footprint data to assess landscape level conservation efforts. Conservation Letters 1:165-172.

Halpern, B. S., C. V. Kappel, K. A. Selkoe, F. Micheli, C. M. Ebert, C. Kontgis, C. M. Crain, R. G. Martone, C. Shearer, and S. J. Teck. 2009. Mapping cumulative human impacts to California Current marine ecosystems. Conservation Letters 2:138-148.

Halpern, B. S., K. A. Selkoe, F. Micheli, and C. V. Kappel. 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology 21:1301-1315.

Halpern, B. S., S. Walbridge, K. A. Selkoe, C. V. Kappel, F. Micheli, C. D'Agrosa, J. F. Bruno, K. S. Casey, C. Ebert, H. E. Fox, R. Fujita, D. Heinemann, H. S. Lenihan, E. M. P. Madin, M. T. Perry, E. R. Selig, M. Spalding, R. Steneck, and R. Watson. 2008. A global map of human impact on marine ecosystems. Science 319:948-952.

Hamilton, G. W. and D. V. Waddington. 1999. Infiltration rates on residential lawns in central Pennsylvania. Journal of Soil and Water Conservation 54:564-568.

Hamilton, P., J. T. Gunn, and G. A. Cannon. 1985. A BOX MODEL OF PUGET-SOUND. Estuarine Coastal and Shelf Science 20:673-692.

Hampton, S. E., P. Romare, and D. E. Seiler. 2006. Environmentally controlled Daphnia spring increase with implications for sockeye salmon fry in Lake Washington, USA. Journal of Plankton Research 28:399-406.

Hansen, A. J., R. L. Knight, J. M. Marzluff, S. Powell, K. Brown, P. H. Gude, and K. Jones. 2005. Effects of exurban development on biodiversity: patterns, mechanisms, and research needs. Ecological Applications 15:1893-1905.

Hansen, M. J. and A. P. Clevenger. 2005. The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. Biological Conservation 125:249-259.

Harrington, T. B. and S. H. Reichard. 2007. Meeting the challenge: invasive plants in Pacific Northwest ecosystems.

Hart Crowser. 2007. Control of Toxic Chemicals in Puget Sound. Phase 1: Initial estimate of loadings. Washington Department of Ecology.

Hartzell, S., A. Leeds, A. Frankel, R. A. Williams, J. Odum, W. Stephenson, and W. Silva. 2002. Simulation of broadband ground motion including nonlinear soil effects for a magnitude 6.5 earthquake on the Seattle fault, Seattle, Washington. Bulletin of the Seismological Society of America 92:831-853.

Harvey, C. J., K. K. Bartz, J. Davies, T. B. Francis, T. P. Good, A. D. Guerry, B. Hanson, K. K. Holsman, J. Miller, M. L. Plummer, J. C. P. Reum, L. D. Rhodes, C. A. Rice, J. F. Samhouri, G. D. Williams, N. Yoder, P. S. Levin, and M. H. Ruckelshaus. 2010. A Mass-balance Model for Evaluating Food Web Structure and Community-scale Indicators in the Central Basin of Puget Sound. NOAA Technical Memorandum NMFS-NWFSC-106, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Seattle, WA.

Hayes, E. H. and W. G. Landis. 2004. Regional ecological risk assessment of a near shore marine environment: Cherry Point, WA. Human and Ecological Risk Assessment 10:299-325.

Heath, M. R. 2008. Comment on "a global map of human impact on marine ecosystems". Science 321.

Hecnar, S. J. and R. T. M'Closkey. 1997. The effects of predatory fish on amphibian species richness and distribution. Biological conservation. 79:123.

Hegerl, G. C., T. R. Karl, M. Allen, N. L. Bindoff, N. Gillett, D. Karoly, X. Zhang, and F. Zwiers. 2006. Climate Change Detection and Attribution: Beyond Mean Temperature Signals. JOURNAL OF CLIMATE 19:5058-5077.

Hepinstall, J. A., M. Alberti, and J. M. Marzluff. 2008. Predicting land cover change and avian community responses in rapidly urbanizing environments. Landscape Ecology 23:1257-1276.

Hepinstall, J. A., J. M. Marzluff, and M. Alberti. 2009. Modeling Bird Responses to Predicted Changes in Land Cover in an Urbanizing Region. Pages 625-659 in J. Millspaugh and F. R. Thompson, editors. Models for Planning Wildlife Conservation in Large Landscapes. Academic Press, San Diego.

Heyerdahl, E. K., D. McKenzie, L. D. Daniels, A. E. Hessl, J. S. Littell, N. J. Mantua, and 3rd. 2008. Climate drivers of regionally synchronous fires in the inland Northwest (1651-1900).

Hidalgo, H. G., T. Das, M. D. Dettinger, D. R. Cayan, D. W. Pierce, T. P. Barnett, G. Bala, A. Mirin, A. W. Wood, C. Bonfils, B. D. Santer, and T. Nozawa. 2009. Detection and Attribution of Streamflow Timing Changes to Climate Change in the Western United States. JOURNAL OF CLIMATE 22:3838-3855.

Hobbs, R. J. 2000. Land-Use Changes and Invasions.in H. A. Mooney and R. J. Hobbs, editors. Invasive species in a changing world. Island Press, Washington, D.C.

Hood, W. G. 2004. Indirect environmental effects of dikes on estuarine tidal channels: Thinking outside of the dike for habitat restoration and monitoring. Estuaries 27:273-282.

Hope, D., W. Zhu, C. Gries, J. Oleson, J. Kaye, N. B. Grimm, and L. A. Baker. 2005. Spatial variation in soil inorganic nitrogen across an arid urban ecosystem. Urban Ecosystems 8:251-273.

Horne, P. J., I. C. Kaplan, K. N. Marshall, P. S. Levin, C. J. Harvey, A. J. Hermann, and E. A. Fulton. 2010. Design and parameterization of a spatially explicit ecosystem model of the Central California Current. NOAA Tech. Memo. NMFS-NWFSC-104, U.S. Department of Commerce, National Oceanic and Atmospheric Administration.

Hulme, P. E. 2009. Trade, transport and trouble: managing invasive species pathways in an era of globalization. Journal of Applied Ecology 46:10-18.

Hutyra, L. R., B. Yoon, and M. Alberti. In press. Terrestrial carbon stocks across a gradient of urbanization: a study of the Seattle, WA region. Global Change Biology 9999.

Hyndman, R. D., S. Mazzotti, D. Weichert, and G. C. Rogers. 2003. Frequency of large crustal earthquakes in Puget Sound-Southern Georgia Strait predicted from geodetic and geological deformation rates. Journal of Geophysical Research-Solid Earth 108.

Iannuzzi, T. J., J. L. Durda, D. V. Preziosi, D. F. Ludwig, R. G. Stahl, Jr., A. A. DeSantis, and R. A. Hoke. 2009. Development of a preliminary relative risk model for evaluating regional ecological conditions in the Delaware River Estuary, USA. Integrated Environmental Assessment and Management 6:164-179.

IPCC. 2007. Climate change 2007 : the physical science basis : contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller, editors. Cambridge University Press, Cambridge; New York.

Ishii, M., M. Kimoto, K. Sakamoto, and S.-I. Iwasaki. 2006. Steric sea level changes estimated from historical ocean subsurface temperature and salinity analyses. Journal of Oceanography 62:155-170.

Jakob, M. and S. Lambert. 2009. Climate change effects on landslides along the southwest coast of British Columbia. Geomorphology 107:275-284.

Jenkins, P. T. and H. A. Mooney. 2006. The United States, China, and invasive species: present status and future prospects.

Jensen, D. 2009. Spill prevention, preparedness and response program. Washington Department of Ecology.

Johannessen, J. and A. MacLennan. 2007. Beaches and Bluffs of Puget Sound. Report No. 2007-04, Puget Sound Nearshore Partnership, Seattle, WA.

Johnson, D. H., M. J. Holloran, J. W. Connelly, S. E. Hanser, C. L. Amundson, and S. T. Knick. 2010. Influences of environmental and anthropogenic features on greater sage-grouse populations, 1997–2007. Studies in Avian Biology: in press.

Johnson, L., T. K. Collier, and J. E. Stein. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAHs) to protect estuarine fish. Aquatic Conserv: Mar. Freshw. Ecosyst. 12:517-538.

Johnson, L. L. and J. T. Landahl. 1993. Contaminant, liver disease, and mortality rates in English sole (Pleuronectes vetulus). Ecology Applications.

Johnson, L. L., J. T. Landahl, L. A. Kubin, B. H. Horness, M. S. Myers, T. K. Collier, and J. E. Stein. 1998. Assessing the effects of anthropogenic stressors on Puget Sound flatfish populations. Journal of Sea Research 39:125-137.

Johnson, L. L., D. P. Lomax, M. S. Myers, O. P. Olson, S. Y. Sol, S. M. O'Neill, J. West, and T. K. Collier. 2008. Xenoestrogen exposure and effects in English sole (Parophrys vetulus) from Puget Sound, WA. Aquatic Toxicology 88:29-38.

Josberger, E. G., W. R. Bidlake, R. S. March, and B. W. Kennedy. 2007. Glacier mass-balance fluctuations in the Pacific Northwest and Alaska, USA. Annals of glaciology. 46:291-296.

Kairis, P. A. and J. M. Rybczyk. 2010. Sea level rise and eelgrass (Zostera marina) production: A spatially explicit relative elevation model for Padilla Bay, WA. Ecological Modelling 221:1005-1016.

Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology and Evolution 21:192-199.

Kenworthy, W. J. and D. E. Haunert, editors. 1991. The light requirements of seagrasses: proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. NOAA Technical Memorandum, NMFS-SEFC-287, National Oceanic and Atmospheric Administration, Beaufort, NC.

Kerr, J. B. and V. E. Fioletov. 2008. Surface Ultraviolet Radiation. Atmosphere-ocean. 46:159.

Kimbrough, K. L., G. G. Lauenstein, J. D. Christensen, and D. A. Apeti. 2008. An assessment of two decades of contaminant monitoring in the Nation's Coastal Zone. NOAA/NOS, Silver Spring, MD.

King County. 2007. Survey of endocrine disruptors in King County surface waters.

King County. 2010. Puget Sound Marine Monitoring: King County Marine and Sediment Assessment Group.

King, K. W. and J. C. Balogh. 2001. Water quality impacts associated with converting farmland and forests to turfgrass. Transactions of the Asae 44:569-576.

Knick, S. T. and S. E. Hanser. 2010. Connecting pattern and process in greater sage-grouse populations and sagebrush landscapes. Studies in Avian Biology: in press.

Koch, F. H., D. Yemshanov, D. W. McKenney, and W. D. Smith. 2009. Evaluating Critical Uncertainty Thresholds in a Spatial Model of Forest Pest Invasion Risk. Risk Analysis 29:1227-1241.

Konrad, C. P., D. B. Booth, and S. J. Burges. 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: Consequences for channel form and streambed disturbance. Water Resources Research 41:W07009, doi:07010.01029/02005WR004097.

Krahn, M. M., M. B. Hanson, R. W. Baird, R. H. Boyer, D. G. Burrows, C. K. Emmons, J. K. B. Ford, L. L. Jones, D. P. Noren, P. S. Ross, G. S. Schorr, and T. K. Collier. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from Southern Resident killer whales. Marine Pollution Bulletin 54:1903-1911.

Krahn, M. M., P. R. Wade, S. T. Kalinowski, M. E. Dalheim, B. L. Taylor, M. B. Hanson, G. M. Ylitalo, R. P. Angliss, J. E. Stein, and R. S. Waples. 2002. Status Review of Southern Resident Killer Whales (Orcinus orca) under the Endangered Species Act. Page 133 in U. S. D. o. Commerce, editor.

Kuldna, P., K. Peterson, H. Poltimae, and J. Luig. 2009. An application of DPSIR framework to identify issues of pollinator loss. Ecological Economics 69:32-42.

Kurz, D. CC, S. G, R. GJ, N. ET, C. AL, E. T, and S. L. 2008. Mountain pine beetle and forest carbon feedback to climate change. Nature 452:987-990.

La Marche, J. L. and D. P. Lettenmaier. 2001. Effects of forest roads on flood flows in the Deschutes River, Washington. Earth Surface Processes and Landforms 26:115-134.

Lackey, R. T. 2003. Pacific Northwest Salmon: Forecasting Their Status in 2100. Reviews in Fisheries Science 11:35-88.

Landis, W. G., P. B. Duncan, E. H. Hayes, and A. J. Markiewicz. 2004. A regional retrospective assessment of the potential stressors causing the decline of the Cherry Point Pacific herring run. Human and Ecological Risk Assessment 10:271-297.

Landis, W. G. and J. K. Wiegers. 2007. Ten years of the relative risk model and regional scale ecological risk assessment. Human and Ecological Risk Assessment 13:25-38.

Langmead, O., A. McQuatters-Gollop, L. D. Mee, J. Friedrich, A. J. Gilbert, M. T. Gomoiu, E. L. Jackson, S. Knudsen, G. Minicheva, and V. Todorova. 2009. Recovery or decline of the northwestern Black Sea: A societal choice revealed by socio-ecological modelling. Ecological Modelling 220:2927-2939.

Larson, E. R. and J. D. Olden. 2008. Do schools and golf courses represent emerging pathways for crayfish invasions? Aquatic Invasions 3:465-468.

Latta, G., H. Temesgen, D. Adams, and T. Barrett. 2010. Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management 259:720-729.

Law, N., L. E. Band, and J. M. Grove. 2004. Nitrogen input from residential lawn care practices in suburban watersheds in Baltimore county, MD. Journal of Environmental Planning and Management 47:737-755.

Lawler, J. and M. Mathias. 2007. Climate Change and the Future of Biodiversity in Washington. Washington Biodiversity Council.

Lawrence, D. J. and J. R. Cordell. 2010. Relative contributions of domestic and foreign sourced ballast water to propagule pressure in Puget Sound, Washington, USA. Biological Conservation In Press, Corrected Proof.

Le Quere, C., M. R. Raupach, J. G. Canadell, G. Marland et al., C. Le Quere et al., C. Le Quere et al., M. R. Raupach, J. G. Canadell, G. Marland, and L. Bopp. 2009. Trends in the sources and sinks of carbon dioxide. NATURE GEOSCIENCE 2:831-836.

Leu, M., S. E. Hanser, and S. T. Knick. 2008. The human footprint in the west: A large-scale analysis of anthropogenic impacts. Ecological Applications 18:1119-1139.

Lichtenstein, S. and P. Slovic. 2006. The Construction of Preference. Cambridge University Press, Cambridge, UK.

Littlell, J. S., D. McKenzie, D. L. Peterson, and A. L. Westerling. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916-2003. Ecological applications : a publication of the Ecological Society of America. 19:1003.

Long, E. R., M. Dutch, S. Aasen, K. Welch, and M. J. Hameedi. 2005. Spatial extent of degraded sediment quality in Puget Sound (Washington State, U.S.A.) based upon measures of the sediment quality triad. Environmental Monitoring and Assessment 111:173-222.

Lubliner, B. 2007. Characterizing stormwater for total maximum daily load studies: a review of current approaches. Washington Department of Ecology, Lacey, WA.

Lubliner, B., M. Redding, and D. Ragsdale. 2010. Pharmaceuticals and personal care products in municipal wastewater and their removal by nutrient treatment technologies. Washington State Department of Ecology, Olympia, WA.

Mangi, S. C., C. M. Roberts, and L. D. Rodwell. 2007. Reef fisheries management in Kenya: Preliminary approach using the driver-pressure-state-impacts-response (DPSIR) scheme of indicators. Ocean & Coastal Management 50:463-480.

Mantua, N. and S. Hare. 2002. The Pacific Decadal Oscillation. Journal of Oceanography 58:35-44.

Marzluff, J. M. 2005. Island biogeography for an urbanizing world: how extinction and colonization may determine biological diversity in human-dominated landscapes. Urban Ecosystems 8:157-177.

Marzluff, J. M., J. C. Withey, K. A. Whittaker, M. D. Oleyar, T. M. Unfried, S. Rullman, and J. DeLap. 2007. Consequences of habitat utilization by nest predators and breeding songbirds across multiple scales in an urbanizing landscape. Condor 109:516-534.

Mato, Y., T. Isobe, H. Takada, H. Kanehiro, C. Ohtake, and T. Kaminuma. 2000. Plastic Resin Pellets as a Transport Medium for Toxic Chemicals in the Marine Environment. Environmental Science & Technology 35:318-324.

Matzen, D. A. and H. B. Berge. 2008. Assessing small-stream biotic integrity using fish assemblages across an urban landscape in the puget sound lowlands of western Washington. Transactions of the American Fisheries Society 137:677-689.

Maxim, L., J. H. Spangenberg, and M. O'Connor. 2009. An analysis of risks for biodiversity under the DPSIR framework. Ecological Economics 69:12-23.

May, C. W., R. R. Horner, J. R. Karr, B. W. Mar, and E. B. Welch. 1997. Effects Of Urbanization On Small Streams in the Puget Sound Ecoregion. Watershed Protection Techniques 2:483-494.

McBride, M. and D. B. Booth. 2005. Urban impacts on physical stream conditions: effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association 41:565-580.

McCarthy, S. G., J. P. Incardona, and N. L. Scholz. 2008. Coastal storms, toxic runoff and the sustainable conservation of fish and fisheries. American Fisheries Society Symposium 64:21.

Meyer, J. L., M. J. Paul, and W. K. Taulbee. 2005. Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society 24:602-612.

Miles, J. R., P. E. Russell, and D. A. Huntley. 2001. Field measurements of sediment dynamics in front of a seawall. Journal of Coastal Research 17:195-206.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Pages 21-48 in S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The Ecology and Management of Wood in World Rivers, American Fisheries Society Symposium 37. American Fisheries Society, Bethesda, MD.

Moore, T. VL, M. NJ, P. MS, L. EA, B. LC, and F. LE. 2008. Impacts of climate variability and future climate change on harmful algal blooms and human health. Environmental health : a global access science source 7.

Moore, C. J. 2008. Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. Environmental Research 108:131-139.

Moore, J. W. 2006. Freshwater invertebrates-red swamp crayfish Procambarus clarkii, Rusty crayfish Orconectes rusticus, Virile crayfish Orconectes virilis.in P. D. Boersma, S. H. Reichard, and A. N. Van Buren, editors. Invasive species in the Pacific Northwest. University of Washington Press, Seattle, WA.

Moore, S. K., N. J. Mantua, J. P. Kellogg, and J. A. Newton. 2008. Local and large-scale climate forcing of. Puget Sound oceanographic properties on seasonal to interdecadal timescales. Limnology and oceanography. 53:1746.

Morgan, M. G. 2003. Characterizing and Dealing With Uncertainty: Insights from the Integrated Assessment of Climate Change. Integrated Assessment 4:46 - 55.

Morgan, R. P. and S. E. Cushman. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. Journal of the North American Benthological Society 24:643-655.

Morley, S. A. and J. R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound basin. Conservation Biology 16:1498-1509.

Mote, P., A. Hamlet, and E. Salathe. 2008. Has spring snowpack declined in the Washington Cascades? HYDROLOGY AND EARTH SYSTEM SCIENCES 12:193-206.

Mote, P. W. 2003. Trends in Temperature and Precipitation in the Pacific Northwest During the Twentieth Century. Northwest science : official publication of the Northwest Scientific Association. 77:271-282.

Mote, P. W., Washington (State). Dept. of Ecology., and University of Washington. Climate Impacts Group. 2008. Sea level rise in the coastal waters of Washington State. Climate Impacts Group ; Dept. of Ecology, State of Washington, [Seattle]; [Olympia].

Mueller, K. W. 2002. First record of the red swamp crayfish, Procambarus clarkii (Girard, 1852) (Decapoda, Cambaridae), from Washington State, USA. Crustaceana 75:94.

Mueller, K. W. 2007. NOTES - STATUS OF THE CRAYFISH STOCKS IN PINE LAKE, KING COUNTY, WASHINGTON FIVE YEARS AFTER THE DISCOVERY OF THE

INVASIVE RED SWAMP CRAYFISH PROCAMBARUS CLARKII (GIRARD, 1852). Journal of freshwater ecology. 22:351.

Muir, D. C. G. and P. H. Howard. 2006. Are There Other Persistent Organic Pollutants? A Challenge for Environmental Chemistsâ€ Environmental Science & Technology 40:7157-7166.

Mumford, T. F. 2007. Kelp and Eelgrass in Puget Sound. Report No. 2007-05, Puget Sound Nearshore Partnership, Seattle, WA.

Murcia, C. 1995. Edge effects in fragmented forests: implications for conservation. Trends in ecology & evolution. 10:58.

Myers, M. S., B. F. Anulacion, B. L. French, W. L. Reichert, C. A. Laetz, J. Buzitis, O. P. Olson, S. Sol, and T. K. Collier. 2008. Improved flatfish health following remediation of a PAH-contaminated site in Eagle Harbor, Washington. Aquatic Toxicology 88:277-288.

Myers, M. S., L. L. Johnson, and T. K. Collier. 2003. Establishing the Causal Relationship between Polycyclic Aromatic Hydrocarbon (PAH) Exposure and Hepatic Neoplasms and Neoplasia-Related Liver Lesions in English Sole (*Pleuronectes vetulus*). Human and Ecological Risk Assessment: An International Journal 9:67 - 94.

Nairn, B., S. Albertson, D. Averill, A. Devol, M. Kawase, J. Newton, C. P. Sarason, and M. Warner. 2005. An Aquatic Biogeochemical Cycling Model Simulation of Puget Sound, WA. Proceedings of the 2005 Puget Sound Georgia Basin Research Conference, Seattle, WA.

Nakawatase, J. and D. Peterson. 2006. Spatial variability in forest growth - climate relationships in the Olympic Mountains, Washington. Canadian Journal of Forest Research 36:77-91.

NatureServe. 2004. NatureServe explorer an online encyclopedia of life. Association for Biodiversity Information and Natural Heritage Network, Arlington, Va.

Natureserve. 2008. NatureServe Web Service, http://services.natureserve.org. Arlington, VA.

Neuman, M., D. St. John, and J. Knauer. 2009. Identification, Definition and Rating of Threats to the Recovery of Puget Sound. Puget Sound Partnership, Olympia, WA.

Newsome, R., N. Tran, G. M. Paoli, L. A. Jaykus, B. Tompkin, M. Miliotis, T. Ruthman, E. Hartnett, F. F. Busta, B. Petersen, F. Shank, J. McEntire, J. Hotchkiss, M. Wagner, and D. W. Schaffner. 2009. Development of a Risk-Ranking Framework to Evaluate Potential High-Threat Microorganisms, Toxins, and Chemicals in Food. Journal of Food Science 74:R39-R45.

Newton, J., T. Mumford, J. Dohrmann, J. West, R. Llanso, H. Berry, and S. Redman. 2000. A conceptual model for environmental monitoring of a marine system. Puget Sound Ambient Monitoring Program, June 2000.

Nightengale, B. J. and C. A. Simenstad. 2001. Overwater Structures: Marine issues. University of Washington, Seattle, Washington.

Nobre, A. M. 2009. An Ecological and Economic Assessment Methodology for Coastal Ecosystem Management. Environmental Management 44:185-204.

O'Neill, S. M. and J. E. West. 2009. Marine Distribution, Life History Traits, and the Accumulation of Polychlorinated Biphenyls in Chinook Salmon from Puget Sound, Washington. Transactions of the American Fisheries Society 138:616-632.

O'Neill, S. M., G. M. Ylitalo, J. E. West, J. Bolton, C. A. Sloan, and M. M. Krahn. 2006. Regional patterns of persistent organic pollutants in five Pacific salmon species (Oncorhynchus spp) and their contributions to contaminant levels in northern and southern resident killer whales (Orcinus orca). Pages 38-42 in Southern Resident Killer Whale Symposium, Seattle, WA.

Ojeda-Martinez, C., F. G. Casalduero, J. T. Bayle-Sempere, C. B. Cebrian, C. Valle, J. L. Sanchez-Lizaso, A. Forcada, P. Sanchez-Jerez, P. Martin-Sosa, J. M. Falcon, F. Salas, M. Graziano, R. Chemello, B. Stobart, P. Cartagena, A. Perez-Ruzafa, F. Vandeperre, E. Rochel, S. Planes, and A. Brito. 2009. A conceptual framework for the integral management of marine protected areas. Ocean & Coastal Management 52:89-101.

Olden, J. D., E. R. Larson, and M. C. Mims. 2009. Home-field advantage: native signal crayfish (Pacifastacus leniusculus) out consume newly introduced crayfishes for invasive Chinese mystery snail (Bellamya chinensis). Aquatic Ecology 43:1073-1084.

Omann, I., A. Stocker, and J. Jager. 2009. Climate change as a threat to biodiversity: An application of the DPSIR approach. Ecological Economics 69:24-31.

Orr, J., V. Fabry, O. Aumont, L. Bopp, S. Doney, R. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, R. Key, K. Lindsay, E. Maier-Reimer, R. Matear, P. Monfray, A. Mouchet, R. Najjar, G. Plattner, K. Rodgers, C. Sabine, J. Sarmiento, R. Schlitzer, R. Slater, I. Totterdell, M. Weirig, Y. Yamanaka, and A. Yool. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature 437:681-686.

OSAC. 2009. Assessment of capacity in Washington State to respond to large-scale marine oil spills: technical report and policy recommendations. Washington Oil Spill Advisory Council.

Ostoich, M., A. Critto, A. Marcomini, E. Aimo, M. Gerotto, and L. Menegus. 2009. Implementation of Directive 2000/60/EC: risk-based monitoring for the control of dangerous and priority substances. Chemistry and Ecology 25:257-275.

Parks, C. G., S. R. Radosevich, B. A. Endress, B. J. Naylor, D. Anzinger, L. J. Rew, B. D. Maxwell, and K. A. Dwire. 2005. Natural and land-use history of the Northwest mountain ecoregions (USA) in relation to patterns of plant invasions. Perspectives in Plant Ecology, Evolution and Systematics 7:137-158.

Parmesan and Y. G. 2003. A globally coherent fingerprint of climate change impacts across natural systems. Nature 421:37-42.

Parson, E. A., P. W. Mote, A. Hamlet, N. Mantua, A. Snover, W. Keeton, E. Miles, D. Canning, and K. G. Ideker. 2001. Potential Consequences of Climate Variability and Change for the Pacific Northwest. Climate change impacts on the United States : the potential consequences of climate variability and change : foundation. Cambridge University Press, Cambridge; New York.

Partridge, V., K. Welch, S. Aasen, and M. Dutch. 2005. Temporal monitoring of Puget Sound sediments: results of the Puget Sound Ambient Monitoring Program, 1989-2000. Washington Department of Ecology.

Patten, M. A. and D. T. Bolger. 2003. Variation in top-down control of avian reproductive success across a fragmentation gradient. Oikos 101:479-488.

Payne, J., J. Bettman, and E. Johnson. 1992. Behavioral decision research: A constructive processing perspective. Annual Review of Psychology 43:87-132.

Pelto, M. S. 2008. Impact of Climate Change on North Cascade Alpine Glaciers, and Alpine Runoff. Northwest science : official publication of the Northwest Scientific Association. 82:65-75.

Penttila, D. 2007. Marine Forage Fishes in Puget Sound. Report No. 2007-03, Puget Sound Nearshore Partnership, Seattle, Washington.

Peterson, A. T. 2003. Predicting the geography of species' invasions via ecological niche modeling. Quarterly Review of Biology 78:419-433.

Peterson, A. T. and C. R. Robins. 2003. Using ecological-niche modeling to predict Barred Owl invasions with implications for Spotted Owl conservation. Conservation Biology 17:1161-1165.

PEW Oceans Commission. 2003. America's Living Oceans: Charting a Course for Sea Change.

Phillips, C. H., K. D. Anderson, and T. O. Ketel. 2008. Spartina Eradication Program 2007 Progress Report. Washington State Department of Agriculture.

Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, P. M. Groffman, L. E. Band, C. G. Boone, W. R. Burch Jr, C. S. B. Grimmond, J. Hom, J. C. Jenkins, N. L. Law, C. H. Nilon, R. V. Pouyat, K. Szlavecz, P. S. Warren, and M. A. Wilson. 2008. Beyond Urban Legends: An Emerging Framework of Urban Ecology, as Illustrated by the Baltimore Ecosystem Study. Bioscience 58:139-150.

Piel, F., M. Gilbert, C. De Cannière, and J.-C. Grégoire. 2008. BIODIVERSITY RESEARCH: Coniferous round wood imports from Russia and Baltic countries to Belgium. A pathway analysis for assessing risks of exotic pest insect introductions. Diversity & Distributions 14:318-328.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and Economic Costs of Nonindigenous Species in the United States. BIOSCIENCE -WASHINGTON- 50:53-65.

Poole, G. C. and C. H. Berman. 2001. An Ecological Perspective on In-Stream Temperature: Natural Heat Dynamics and Mechanisms of Human-Caused Thermal Degradation. ENVIRONMENTAL MANAGEMENT -NEW YORK- 27:787-802.

Poston, T. 2001. Treated Wood Issues Associated with Overwater Structures in Marine and Freshwater Environments. Prepared for the Washington Department of Fish and Wildlife, Washington Department of Ecology, and Washington Department of Transportation.

Potschin, M. 2009. Land use and the state of the natural environment. Land Use Policy 26:S170-S177.

PRISM. 2010. The Puget Sound Regional Synthesis Model Initiative, http://www.prism.washington.edu/home/index. University of Washington, Seattle, WA.

Quinn, T. 1997. Coyote (Canis latrans) habitat selection in urban areas of western Washington via analysis of routine movements. Northwest Science 71:289-297.

Quinn, T. 1997. Coyote (Canis latrans) food habits in three urban habitat types of western Washington. Northwest Science 71:1-5.

Ragab, R., J. Bromley, P. Rosier, J. D. Cooper, and J. H. C. Gash. 2003. Experimental study of water fluxes in a residential area: 1. Rainfall, roof runoff and evaporation: the effect of slope and aspect. Hydrological Processes 17:2409-2422.

Ragab, R., P. Rosier, A. Dixon, J. Bromley, and J. D. Cooper. 2003. Experimental study of water fluxes in a residential area: 2. Road infiltration, runoff and evaporation. Hydrological Processes 17:2423-2437.

Ramsay, M., N. Salisbury, and S. Surbey. 2004. A Citywide Survey of Habitats on Public Land in Seattle, a tool for urban restoration planning and ecological monitoring.in 16th International Conference, Society for Ecological Restoration, Victoria, Canada.

Randall, J. M., L. E. Morse, N. Benton, R. Hiebert, S. Lu, and T. Killeffer. 2008. The Invasive Species Assessment Protocol: A Tool for Creating Regional and National Lists of Invasive Nonnative Plants That Negatively Impact Biodiversity. Invasive Plant Science and Management 1:36-49.

Reichard, S. H. and P. White. 2001. Horticulture as a Pathway of Invasive Plant Introductions in the United States. Bioscience 51:103-113.

Resources, W. D. o. N. 2007. State of Washington Natural Heritage Plan 2007. Washington Department of Natural Resources, Olympia, WA.

Rice, C. 2006. Effects of shoreline modification on a Northern Puget Sound beach: Microclimate and embryo mortality in surf smelt (Hypomesus pretiosus). Estuaries and Coasts 29:63-71.

Richter, A. and S. Kolmes. 2005. Maximum Temperature Limits for Chinook, Coho, and Chum Salmon, and Steelhead Trout in the Pacific Northwest. Reviews in Fisheries Science 13:23-49.

Richter, K. O. and A. L. Azous. 1995. Amphibian occurrence and wetland characteristics in the Puget Sound Basin. Wetlands : the journal of the Society of the Wetlands Scientists.:305.

Roberts, M. L. and R. E. Bilby. 2009. Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams. Journal of the North American Benthological Society 28:941-954.

Robinson, L., J. P. Newell, and J. M. Marzluff. 2005. Twenty-five years of sprawl in the Seattle region: growth management responses and implications for conservation. Landscape and Urban Planning 71:51-72.

Root, T. L., J. T. Price, K. R. Hall, S. H. Schneider, C. Rosenzweig, and J. A. Pounds. 2003. Fingerprints of global warming on wild animals and plants. 421:57-60.

Ross, P. S. 2006. Fireproof killer whales (Orcinus orca): flame retardant chemicals and the conservation imperative in the charismatic icon of British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Science 63:224-234.

Ross, P. S., G. M. Ellis, M. G. Ikonomou, L. G. Barrett-Lennard, and R. F. Addison. 2000. High PCB concentrations in free-ranging Pacific killer whales, Orcinus orca: effects of age, sex and dietary preference. Marine Pollution Bulletin 40:504-515.

Roura-Pascual, N., D. M. Richardson, R. M. Krug, A. Brown, R. A. Chapman, G. G. Forsyth, D. C. Le Maitre, M. P. Robertson, L. Stafford, B. W. Van Wilgen, A. Wannenburgh, and N. Wessels. 2009. Ecology and management of alien plant invasions in South African fynbos: Accommodating key complexities in objective decision making. Biological Conservation 142:1595-1604.

Roy, A. H., M. C. Freeman, B. J. Freeman, S. J. Wenger, W. E. Ensign, and J. L. Meyer. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. Journal of the North American Benthological Society 24:656-678.

Roy, A. H., A. H. Purcell, C. J. Walsh, and S. J. Wenger. 2009. Urbanization and stream ecology: five years later. Journal of the North American Benthological Society 28:908-910.

Ruckelshaus, M. H., M. M. McClure, and coordinators. 2007. Sound Science: Synthesizing Ecological and Socioeconomic Information about the Puget Sound Ecosystem. U.S. Dept. of Commerce, National Oceanic & Atmospheric Administration (NMFS), Northwest Fisheries Science Center; prepared in cooperation with the Sound Science collaborative team, Seattle, WA.

Ruesink, J. L., H. S. Lenihan, A. C. Trimble, K. W. Heiman, F. Micheli, J. E. Byers, and M. C. Kay. 2005. Introduction of non-native oysters: Ecosystem effects and restoration implications. Annual Review of Ecology Evolution and Systematics 36:643-689.

Ruiz, G. M. and J. T. Carlton. 2003. Invasive species : vectors and management strategies. Island Press, Washington, DC.

Salathé, E. P., R. Steed, C. F. Mass, and P. H. Zahn. 2008. A High-Resolution Climate Model for the U.S. Pacific Northwest: Mesoscale Feedbacks and Local Responses to Climate Change. JOURNAL OF CLIMATE 21:5708-5726.

Samhouri, J. F., P. S. Levin, and C. J. Harvey. 2009. Quantitative Evaluation of Marine Ecosystem Indicator Performance Using Food Web Models. Ecosystems 12:1283-1298.

Sanderson, E. W., M. Jaiteh, M. A. Levy, K. H. Redford, A. V. Wannebo, and G. Woolmer. 2002. The human footprint and the last of the wild. Bioscience 52:891-904.

Scheuerell, M. D. and D. E. Schindler. 2003. Diel vertical migration by juvenile sockeye salmon: Empirical evidence for the antipredation window. Ecology 84:1713-1720.

Schlenger, P., A. MacLennan, E. Iverson, K. Fresh, C. Tanner, B. Lyons, S. Todd, R. Carman, D. Myers, S. Campbell, and A. Wick. In review. Strategic Needs Assessment Report. Prepared for the Puget Sound Nearshore Ecosystem Restoration Project.

Schmidt-Etkin, D. 2009. Oil spill risk in industry sectors regulated by Washington State Department of Ecology Spills Program for Oil Spill Prevention and Preparedness. Washington Department of Ecology.

Schooler, S. S., P. B. McEvoy, P. Hammond, and E. M. Coombs. 2009. Negative per capita effects of two invasive plants, Lythrum salicaria and Phalaris arundinacea, on the moth diversity of wetland communities. Bulletin of Entomological Research 99:229-243.

Schueler, T. R. 1995. The peculiarities of perviousness. Watershed Protection Techniques 2:233-238.

Seattle Urban Nature Project. 2006. Citywide Habitat Assessment: Interim Report. Seattle, WA.

Selkoe, K. A., B. S. Halpern, and R. J. Toonen. 2008. Evaluating anthropogenic threats to the Northwestern Hawaiian Islands. Aquatic Conservation-Marine and Freshwater Ecosystems 18:1149-1165.

Semmens, B. 2008. Acoustically derived fine-scale behaviors of juvenile Chinook salmon (Oncorhynchus tshawytscha) associated with intertidal benthic habitats in an estuary. Canadian Journal of Fisheries and Aquatic Sciences 65:2053-2062.

Shandas, V. and M. Alberti. 2009. Exploring the role of vegetation fragmentation on aquatic conditions: Linking upland with riparian areas in Puget Sound lowland streams. Landscape and Urban Planning 90:66-75.

Shipman, H. 2008. A Geomorphic Classification of Puget Sound Nearshore Landforms. Report No. 2008-01, Puget Sound Nearshore Partnership, Seattle, Washington.

Simenstad, C. A., M. Ramirez, J. Burke, M. Logsdon, H. Shipman, C. Tanner, C. Davis, J. Fung, P. Bloch, K. Fresh, D. Myers, E. Iverson, A. Bailey, P. Schlenger, C. Kiblinger, P. Myre, W. Gerstel, and A. MacLennan. 2009. Historic Change of Puget Sound Shorelines: Puget Sound Nearshore Ecosystem Project Change Analysis. Puget Sound Nearshore Report No. 2009-XX, Published by Washington Department of Fish and Wildlife, Olympia, Washington, and U.S. Army Corps of Engineers, Seattle, Washington.

Simkanin, C., I. Davidson, M. Falkner, M. Sytsma, and G. Ruiz. 2009. Intra-coastal ballast water flux and the potential for secondary spread of non-native species on the US West Coast. Marine Pollution Bulletin 58:366-374.

Simonich, S. L. and R. A. Hites. 1995. Global Distribution of Persistent Organochlorine Compounds. Science 269:1851-1854.

Skerratt, L. F., L. Berger, R. Speare, S. Cashins, K. R. McDonald, A. D. Phillott, H. B. Hines, and N. Kenyon. 2007. Spread of chytridiomycosis has caused the rapid global decline and extinction of frogs. Ecohealth 4:125-134.

Smith, R. F., L. C. Alexander, and W. O. Lamp. 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. Journal of the North American Benthological Society 28:1022-1037.

Snow, J. A., J. B. Dennison, D. A. Jaffe, H. U. Price, J. K. Vaughan, and B. Lamb. 2003. Aircraft and surface observations of air quality in Puget Sound and a comparison to a regional model. Atmospheric Environment 37:4019-4032.

Sobocinski, K. L. 2003. The Impact of Shoreline Armoring on Supratidal Beach Fauna of Central Puget Sound. Master of Science thesis, University of Washington, Seattle, Washington.

Spangenberg, J. H., J. Martinez-Alier, I. Omann, I. Monterroso, and R. Binimelis. 2009. The DPSIR scheme for analysing biodiversity loss and developing preservation strategies. Ecological Economics 69:9-11.

Spromberg, J. A. and J. P. Meador. 2006. Relating chronic toxicity responses to population-level effects: A comparison of population-level parameters for three salmon species as a function of low-level toxicity. Ecological Modelling 199:240-252.

Steen, D. A. and J. P. Gibbs. 2004. Effects of Roads on the Structure of Freshwater Turtle Populations. Conservation Biology 18:1143-1148.

Steuer, J. J., J. D. Bales, and E. M. P. Giddings. 2009. Relationship of stream ecological conditions to simulated hydraulic metrics across a gradient of basin urbanization. Journal of the North American Benthological Society 28:955-976.

Stewart, I. T., D. R. Cayan, and M. D. Dettinger. 2005. Changes toward Earlier Streamflow Timing across Western North America. JOURNAL OF CLIMATE 18:1136-1155.

Stinson, D. W. 2005. Washington State status report for the Mazama pocket gopher, streaked horned lark, and Taylor's checkerspot. Washington Department of Fish and Wildlife, Olympia, WA.

Stoelinga, M. T., M. D. Albright, and C. F. Mass. 2009. A New Look at Snowpack Trends in the Cascade Mountains. Journal of Climate- Submitted unpublished.

Sumpter, J. P. and A. C. Johnson. 2005. Lessons from Endocrine Disruption and Their Application to Other Issues Concerning Trace Organics in the Aquatic Environment. Environmental Science & Technology 39:4321-4332.

Tague, C. L. and L. E. Band. 2001. Simulating the impact of road construction and forest harvesting on hydrologic response. Earth Surface Processes and Landforms 26:135-151.

Tague, C. L. and L. E. Band. 2004. RHESSys: Regional Hydro-Ecologic Simulation System -- An Object-Oriented Approach to Spatially Distributed Modeling of Carbon, Water, and Nutrient Cycling. Earth Interactions 8:1-42.

Teck, S. J., B. S. Halpern, C. V. Kappel, F. Micheli, K. A. Selkoe, C. M. Crain, R. Martone, C. Shearer, J. Arvai, B. Fischhoff, G. Murray, R. Neslo, and R. Cooke. 2010. Using expert judgment to estimate marine ecosystem vulnerability in the California Current. Ecological Applications 20:1402-1416.

Theobald, D. M. 2010. Estimating natural landscape changes from 1992 to 2030 in the conterminous US. Landscape Ecology: published on-line.

Thom, R. M. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. Wetlands 12:147-156.

Thom, R. M., G. D. Williams, J. D. Toft, S. L. Southard, C. W. May, G. A. McMichael, J. A. Vucelick, J. T. Newell, and J. A. Southard. 2006. Impacts of Ferry Terminals and Ferry Operations on Juvenile Salmon Migrating along Puget Sound Shorelines. Prepared for the Washington State Department of Transportation. Prepared by Battelle Memorial Institute, Pacific Northwest Division.

Thompson, R. C., Y. Olsen, R. P. Mitchell, A. Davis, S. J. Rowland, A. W. G. John, D. McGonigle, and A. E. Russell. 2004. Lost at Sea: Where Is All the Plastic? Science 304:838-.

Thomson, R. E. 1994. Physical oceanography of the Strait of Georgia-Puget Sound-Juan de Fuca Strait system. Pages 36-98 in Review of the Marine Environment and Biota of Strait of Georgia, Puget Sound, and Juan de Fuca Strait: Proceedings of the BC/Washington Symposium on the Marine Environment, Jan 13&14, 1994. Can. Tech. Rep. Fish. Aquat. Sci. Report No.1948. 398 pp.

Tonnes, D. M. 2008. Ecological functions of marine riparian areas and driftwood along north Puget Sound shorelines. M.S. Thesis. University of Washington, Seattle, WA.

Toran, L. and D. Grandstaff. 2007. Variation of Nitrogen Concentrations in Stormpipe Discharge in a Residential Watershed. JAWRA Journal of the American Water Resources Association 43:630-641.

Trimble, A. C., J. L. Ruesink, and B. R. Dumbauld. 2009. Factors preventing the recovery of a historically overexploited shellfish species, Ostrea lurida Carpenter 1864. JOURNAL OF SHELLFISH RESEARCH 28:97-106.

Tsai, H. T., S. Y. Tzeng, H. H. Fu, and J. C. T. Wu. 2009. Managing multinational sustainable development in the European Union based on the DPSIR framework. African Journal of Business Management 3:727-735.

Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. Landscape Ecology in Theory and Practice: Pattern and Process. Springer-Verlag, New York, NY.

UNFPA. 2007. State of World Population 2007: Unleashing the Potential of Urban Growth. United Nations Population Fund.

Urgenson, L. S., S. H. Reichard, and C. B. Halpern. 2009. Community and ecosystem consequences of giant knotweed (Polygonum sachalinense) invasion into riparian forests of western Washington, USA. Biological Conservation 142:1536-1541.

Valiela, I. and J. L. Bowen. 2002. Nitrogen sources to watersheds and estuaries: role of land cover mosaics and losses within-watersheds. Environmental Pollution 118:239-248.

Valle, S., M. A. Panero, and L. Shor. 2007. Pollution Prevention and Management Strategies for Polycyclic Aromatic Hydrocarbons in the New York/New Jersey Harbor. Harbor Consortium, New York Academy of Sciences, New York, NY.

van Kleunen, M., W. E, and F. M. 2010. A meta-analysis of trait differences between invasive and non-invasive plant species. Ecology Letters 13:235-245.

Vaughan, J., B. Lamb, C. Frei, R. Wilson, C. Bowman, C. Figueroa-Kaminsky, S. Otterson, M. Boyer, C. Mass, M. Albright, J. Koenig, A. Collingwood, M. Gilroy, and N. Maykut. 2004. A numerical daily air quality forecast system for the Pacific Northwest. Bulletin of the American Meteorological Society 85:549-561.

Veldhoen, N., M. G. Ikonomou, C. Dubetz, N. MacPherson, T. Sampson, B. C. Kelly, and C. C. Helbing. 2010. Gene expression profiling and environmental contaminant assessment of migrating Pacific salmon in the Fraser River watershed of British Columbia. Aquatic Toxicology In Press, Corrected Proof.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, and G. D. Tilman. 1997. Human alteration of the global nitrogen cycle: Sources and consequences. Ecological Applications 7:737-750.

Volk, C. J., P. M. Kiffney, and R. L. Edmonds. 2003. Role of riparian red alder (Alnus rubra) in the nutrient dynamics of coastal streams of the Olympic Peninsula, WA, U.S.A. American Fisheries Society Special Publication 34:213-228.

Waddell, P., A. Borning, M. Noth, N. Freier, and M. Becke. 2003. Microsimulation of urban development and location choices: design and implementation of UrbanSim. Networks and Spatial Economics 3:43-67.

Walsh, C. J., T. D. Fletcher, and A. R. Ladson. 2005. Stream restoration in urban catchments through redesigning stormwater systems: looking to the catchment to save the stream. Journal of the North American Benthological Society 24:690-705.

Walsh, C. J. and J. Kunapo. 2009. The importance of upland flow paths in determining urban effects on stream ecosystems. Journal of the North American Benthological Society 28:977-990.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan, II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706-723.

Washington Department of Ecology. 2010. Control of Toxic Chemicals in Puget Sound.

Washington Invasive Species Council. 2009. Annual Report. Seattle.

WBC. 2007. Washington's Biodiversity: Status and Threats. Washington Biodiversity Council, Olympia, WA.

WDFW. 2009. Landscape Planning for Washington's Wildlife: Managing Biodiversity in Developing Areas. Washington Department of Fish and Wildlife, Olympia, WA.

Weller, D. E., T. E. Jordan, D. L. Correll, and Z. J. Liu. 2003. Effects of land-use change on nutrient discharges from the Patuxent River watershed. Estuaries 26:244-266.

Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Marti, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramirez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080-1098.

West, J. E., S. M. O'Neill, G. R. Lippert, and S. R. Quinnell. 2001. Toxic contaminants in marine and anadromous fish from Puget Sound, Washington: Results from the Puget Sound Ambient Monitoring Program Fish Component, 1989-1999. Technical Report FTP01-14, Washington Department of Fish and Wildlife, Olympia, WA.

West, J. E., S. M. O'Neill, and G. M. Ylitalo. 2008. Spatial extent, magnitude, and patterns of persistent organochlorine pollutants in Pacific herring (Clupea pallasi) populations in the Puget Sound (USA) and Strait of Georgia (Canada). Science of The Total Environment 394:369-378.

Westerling, H. HG, C. DR, and S. TW. 2006. Warming and earlier spring increase western U.S. forest wildfire activity. Science (New York, N.Y.) 313:940-943.

Westphal, M., M. Browne, K. MacKinnon, and I. Noble. 2008. The link between international trade and the global distribution of invasive alien species. Biological Invasions 10:391-398.

Whitaker, A., Y. Alila, J. Beckers, and D. Toews. 2003. Application of the distributed hydrology soil vegetation model to redfish creek, British columbia: model evaluation using internal catchment data. Hydrological Processes 17:199-224.

White, E. M., A. T. Morzillo, and R. J. Alig. 2009. Past and projected rural land conversion in the US at state, regional, and national levels. Landscape and Urban Planning 89:37-48.

Whittaker, K. A. and J. M. Marzluff. 2009. Species-specific survival and relative habitat use in an urban landscape during the postfledging period. Auk 126:288-299.

Wickham, J. D., R. V. O'Neill, K. H. Riitters, E. R. Smith, T. G. Wade, and K. B. Jones. 2002. Geographic targeting of increases in nutrient export due to future urbanization. Ecological Applications 12:93-106.

Wiegand, T., E. Revilla, and K. A. Moloney. 2005. Effects of habitat loss and fragmentation on population dynamics. Conservation Biology 19:108-121.

Wigmosta, M. S., B. Nijssen, and P. Storck. 2002. The Distributed Hydrology Soil Vegetation Model. Pages 7-42 in V. P. Singh and D. Frevert, editors. Mathematical Models of Small Watershed Hydrology and Applications. Water Resources Publications, LLC, Highlands Ranch, CO.

Wigmosta, M. S., L. W. Vail, and D. P. Lettenmaier. 1994. A distributed hydrology-vegetation model for complex terrain. Water Resources Research 30:1665-1679.

Williams, D. W. and A. M. Liebhold. 2002. Climate change and the outbreak ranges of two North American bark beetles. Agricultural & Forest Entomology 4:87-99.

Williams, S. L. 2007. Introduced species in seagrass ecosystems: Status and concerns. Journal of Experimental Marine Biology and Ecology 350:89-110.

Winder, M. and D. E. Schindler. 2004. Climatic effects on the phenology of lake processes. Global Change Biology 10:1844-1856.

With, K. A. 2002. The Landscape Ecology of Invasive Spread. Conservation Biology 16:1192-1203.

Withey, J. C. and J. M. Marzluff. 2009. Multi-scale use of lands providing anthropogenic resources by American Crows in an urbanizing landscape. Landscape Ecology 24:281-293.

WOFM. 2007. Washington State County Growth Management Population Projections: 2000 to 2030. Washington State Office of Financial Management, Olympia, WA.

WOFM. 2010. 2009 Data Book. Forecasting Division, State of Washington Office of Financial Management, Olympia, WA.

Wootton, J. T., C. A. Pfister, and J. D. Forester. 2008. Dynamic patterns and ecological impacts of declining ocean pH in a high-resolution multi-year dataset. Proceedings of the National Academy of Sciences of the United States of America. 105:18848.

Zhang, X., L. A. Vincent, W. D. Hogg, and A. Niitsoo. 2000. Temperature and Precipitation Trends in Canada During the 20th Century. Atmosphere-ocean. 38:395.

Zhou, W., X. Tang, H. Xiao, Y. Wang, and R. Wang. 2009. Response of marine microalgae, heterotrophic bacteria and their relationship to enhanced UV-B radiation. J. Ocean Univ. China Journal of Ocean University of China 8:35-38.

Zhu, W. X. and M. A. Carreiro. 2004. Temporal and spatial variations in nitrogen transformations in deciduous forest ecosystems along an urban - rural gradient. Soil Biology & Biochemistry 36:267-278.

Zickfeld, K., M. Granger Morgan, D. J. Frame, and K. D.W. 2010. Expert judgments about transient climate response to alternative future trajectories of radiative forcing. Proceedings of the National Academy of Sciences, in press.

Zolbrod, A. and D. Peterson. 1999. Response of high-elevation forests in the Olympic Mountains to climatic change. Canadian Journal of Forest Research 29:1966-1978.

Chapter 4. Ecosystem Protection and Restoration Strategies

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Overarching, Large-Scale Protection and Restoration Strategies: John Lombard[2]
Strategies for Watersheds and Tributaries: Richard R. Horner[3]
Strategies for Marine and Estuarine Habitats: E. Eric Knudsen[1]
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Section 1. Introduction

The goal of this chapter is to review the potential ecosystem protection and restoration strategies investigated in past scientific research, assess how they can positively affect the biophysical condition of the greater Puget Sound ecosystem and summarize how the strategies can be applied to reduce threats to recovery of the Puget Sound ecosystem. This chapter covers strategies for both protecting resources that remain healthy as well as rehabilitating impaired natural resources. We emphasize the importance of concentrating on determining the level of effectiveness of the candidate strategies based on scientific research, as well as the relative certainty associated with their reported effectiveness.

We reviewed the background and evaluated the relative scientific basis for the effectiveness of the most promising and well-substantiated strategies as well as relevant strategies that hold promise for the future. We included placeholders for both established and future strategies that were not covered. Socioeconomic strategies for Puget Sound ecosystem protection and restoration were touched upon briefly but can be expanded in future iterations of the Puget Sound Science Update.

We particularly focus on identifying strategies that reduce multiple threats to the ecosystem by linking the strategies to their threat reduction objectives under the description of each strategy. Although we do not make recommendations for the application of certain strategies relative to others, we do include a proposed evaluation process that can be used as a to compare attributes and relative cost-effectiveness of different strategies.

We define a protection and restoration strategy as any action that will protect, restore, or improve the functional well-being of the natural Puget Sound ecosystem. Identifying a strategy requires identifying a goal or goals, identifying possible actions (choices) to achieve the goal, evaluating the likely success of those actions, and deciding on a relatively complete set of actions.

Protection and restoration strategies are strongly characterized by elements of variable scale (e.g., geographic, institutional, temporal), complexity, technical application and degrees of overlap.

1. Organization of Chapter 4 of the Puget Sound Science Update

Because of the complex, dynamic, and interconnected nature of ecosystems and how they interrelate with human institutional systems and practical aspects of physical, on-the-ground application, protection and restoration strategies do not fall into neat categories. Therefore our chapters are organized according to how the strategies will be implemented. First, in Section 2 we address the overarching principles for protection and restoration strategies and review broad strategies that, by their nature, apply generally across the landscape, such as land protection and flow protection. In Section 3 we review protection and restoration strategies that apply to the physical, chemical, and ecological functions of streams, tributaries, and watershed habitat quality. We address in Section 4 strategies that directly influence the ecology and habitats of Puget Sound proper, its estuaries, and shorelines. In Section 5, we review strategies that directly apply to the recovery of fish and wildlife populations. In each section, we provide background regarding the strategy, its application in Puget Sound, and its scientifically supportable effectiveness, recognizing the multitude of strategies and topics that were not covered in this first iteration of the PSSU.

A systematic approach is required for decision-makers to understand the relationships among different types and scales of protection and restoration strategies and for gauging the effectiveness of the various strategies. There is also a need distinguish among strategies that already known to be effective, those that need additional research and those for which there is promise but little information. Therefore, in Section 6 we propose a system for organizing, and ultimately rating different strategies.

We do not address specific implementation or monitoring requirements. On-site applications of protection and restoration measures are decided at the federal, state, tribal, and local levels. Importantly, systems for monitoring the relative success of various protection and restoration strategies must be implemented to provide an information feedback loop needed to evaluate relative success of the measures.

Overarching, Large-scale Protection and Restoration Strategies

Here we focus on strategies that address broad-scale impacts in Puget Sound. We discuss perhaps the two must ubiquitous drivers, human footprint and climate change, recognizing that all other strategies must be imbedded within the context of these ultimate drivers. This review concentrates on publications that focus on Puget Sound, or at least the Pacific Northwest, including: Clancy et al. (2009), Climate Impacts Group (2009), Hulse, Gregory, and Baker (2002), Lombard (2006), Montgomery et al. (2003), and Ruckelshaus and McClure (2007). It is our hope that future versions of this document include lessons learned from other large-scale protection and restoration efforts in the U.S. that have analogous processes or properties.

1. Ultimate Drivers: Human Footprint and Climate Change

In coming decades, the key drivers of ecological change for the Puget Sound ecosystem will be the likely increasing size of the human footprint (a function of both the region's growing population and per capita impacts) and climate change. To acknowledge these external driving factors, we propose that the overall strategy to protect and restore the Puget Sound ecosystem be guided by three broad principles:

1. Many valuable mitigation actions address impacts from both human footprint and future climate. Many of the most valuable actions to mitigate the impacts of climate change are also among the most valuable actions to reduce per capita impacts of the human footprint; the rationale for action therefore does not depend on predictions of climate change, but is strengthened by the potential to provide multiple benefits (Whitely Binder et al. 2009).
2. Increasing resilience of the ecosystem will allow ecological functions to continue in the face of climate change, increased weather extremes and other stressors (Whitely Binder et al. 2009).
3. Principles of adaptive management are important components of protection and restoration actions in general.

To address the threats posed by climate change (See Chapter 3 of the PSSU), specific actions for Puget Sound proposed by the Climate Impacts Group (CIG 2009) could improve ecological functioning and increase the resilience of the ecosystem to other stresses from regional population growth. These actions are included in the PSP Action Agenda (PSP 2008), and include reducing water demand, restoring riparian areas, protecting and restoring off-channel habitats in floodplains, maximizing stormwater infiltration, expanding or adjusting protected areas to induce greater habitat and climatic diversity to permit successful shifts in species distributions and prevent new development on beaches and bluffs likely to be threatened by sea level rise.

Puget Sound Protection and Restoration Strategies

At the landscape scale, the priority strategies identified in the PSP Action Agenda include those from the Puget Sound Salmon Recovery Plan for the watersheds, estuaries and nearshore habitats, and fall under 4 main categories: Protection of intact ecosystem processes, restoration of ecosystem processes that are no longer intact, prevention of water pollution at its source and working together as a coordinated system (Shared Strategy 2007, PSP 2008).

General principles for implementing site-specific protection and restoration strategies include understanding the physical setting for the proposed action (Buffington et al. 2003, Bolton et al. 2003), prioritizing protection of highly functioning habitats over restoration of damaged ones, focusing on both the protection and restoration of habitat forming processes and connectivity (Clancy et al. 2009) and treating protection and restoration actions as experiments with explicit, testable hypotheses and monitoring to assess their effectiveness.

Landscape protection strategies can include two different approaches: focusing growth away from ecologically important and sensitive areas, and permanently protecting intact areas that still function well, both of which are included in the PSP's approach (Neuman et al. 2009). The strategy of focusing growth away from ecologically important and sensitive areas was found to be important for achieving ecological goals over longer (50 year) time scales by Hulse et al. (2002) in the Willamette River basin. Parametrix (2003) also found that this strategy was critical for achieving ecological goals over long timeframes at smaller scales, such as the 16-square-mile basin of Chico Creek, on the Kitsap Peninsula. Both studies found that growth in rural areas—how much occurs and where—was particularly important for the larger ecosystem.

As a supplement to the strategy of permanently protecting areas that still function well, the literature supports including key areas for habitat-forming processes, even those which are not currently intact. Acquisition of property is an effective strategy for permanent protection (Clancy et al. 2009), however it not always feasible across the entire scale where protection and restoration are needed. Additional strategies, including restoration and regulation can supplement the benefits of protection and achieve ecological functioning across larger scales (Lombard 2006).

Funding large scale ecosystem restoration

Numerous studies have stressed the importance of a stable source of funding for large-scale ecosystem restoration. Adler, Michele, and Green (2000) state that "funding stability is as important as absolute funding levels." They go on to suggest that "Congress should consider establishing longer-term funding arrangements for watershed and other environmental programs that must be designed and implemented over long periods of time." Similarly, NRC (2008) found that "The executive and legislative branches of the federal government should consider departing from traditional project-by-project review, authorization and yearly funding to benefit both the [Everglades project] and other multi-component ecosystem restoration projects across the nation." The Everglades project appears to be a particularly acute cautionary example warning against too great a dependence on the federal government for support of large-scale restoration. Beyond problems caused by delays and unpredictability in federal authorizations, NRC (2008) found that "... the most serious cause ..." of overall delays was the "... complex and lengthy…planning and authorization process ..." mandated by Congress for each individual project.

Economics

The broad definition of "regulation" in Montgomery, Booth, and Bolton (2003) includes incentives, noting that incentives are intended to address conflicts between public costs or benefits and those of private decision-makers. More generally, these are instances of what economists call an "externality, " which is when a purchase or use decision by one set of parties has effects on others who do not have a choice in the decision and whose interests are therefore generally not taken into account. Economic production that pollutes air or water is the classic example of a negative environmental externality, but externalities can also be positive. Agricultural practices certified as "Salmon-Safe," for example, benefit water quality and salmon populations (and, therefore, the wider public that values them), yet the public provides no compensation for these benefits. The predictable result is that too much economic activity occurs with negative externalities and too little with positive externalities.

A.C. Pigou is generally recognized as the first major economist to grapple seriously with the problem of internalizing environmental externalities (Pigou 1932—originally published in 1920). Today, the fields of resource, environmental and ecological economics all address this problem, although from different perspectives (Tietenberg 2006; Daly and Farley 2004). Pigou (1932) proposed a tax equal to the marginal external cost as the seemingly simple solution to the problem of negative environmental externalities. However, identifying an actual value for marginal external costs is extremely challenging, even though numerous methodologies have been developed to do so (see, for example, Freeman 2003)[1] . Ecological economics argues that instead of spending enormous efforts to calculate the "correct" value of negative or positive environmental externalities, we should act on our knowledge that the price of zero currently attributed to them is incorrect and work to implement and improve policies that address this key deficiency (Daly and Farley 2004).

In the Puget Sound area, Lombard (2006) suggests the possibility of using a tax or fee to address negative environmental externalities from water withdrawals, our transportation system, discharge of pollutants, and landscape-scale environmental consequences from growth. He also suggests that revenues could help fund programs to reward landowners for ecological services from their land that currently go uncompensated. This could include providing more "space" for rivers, the nearshore, and other key places on the landscape for ecological processes. No analysis has been performed to estimate the effects these taxes and fees would likely have on the amount of these activities.

[1] More details on economic valuation of ecosystem services await the separate chapter on socio-economic strategies expected to be added to this section in a future update.

References

Adler, R., S. Michele, and H. Green. 2000. Lessons from Large Watershed Programs: A Comparison of the Colorado River Basin Salinity Control Program with the San Francisco Bay-Delta Program, Central and South Florida, and the Chesapeake Bay Program. National Academy of Public Administration, Center for the Economy and the Environment, Washington, DC.

Bolton, S., D.B. Booth, and D.R. Montgomery.2003. Restoration of Puget Sound Rivers: Do We Know How to Do It? In D.B. Montgomery, S. Bolton, D.R. Booth, and L. Wall (eds.), Restoration of Puget Sound Rivers, University of Washington Press, Seattle, WA, pp.483-490.

Buffington, J.M., R.D. Woodsmith, D.B. Booth, and D.R. Montgomery. 2003. Fluvial Processes in Puget Sound Rivers and the Pacific Northwest. In D.B. Montgomery, S. Bolton, D.R. Booth, and L. Wall (eds.), Restoration of Puget Sound Rivers, University of Washington Press, Seattle, WA, pp.46-78.

Clancy, M., I. Logan, J. Lowe, J. Johannessen, A. MacLennan, F.B. Van Cleve, J. Dillon, B. Lyons, R. Carman, P. Cereghino, B. Barnard, C. Tanner, D. Myers, R. Clark, J. White, C. Simenstad, M. Gilmer, and N. Chin. 2009. Management Measures for Protecting and Restoring the Puget Sound Nearshore. Technical Report 2009-01. Prepared in support of the Puget Sound Nearshore Ecosystem Restoration Project, Seattle, WA.

Clark, W., R. Mitchell, D. Cash, F. Alcock.2002. Information as Influence: How Institutions Mediate the Impact of Scientific Assessments on Global Environmental Affairs. John F. Kennedy School of Government, Harvard University, Cambridge, MA.

Climate Impacts Group. 2009. The Washington Climate Change Impacts Assessment. M.M Elsner, J. Littell, and L.W. Binder (eds.). Center for Science in the Earth System, Joint Institute for the Study of Atmosphere and Oceans, University of Washington, Seattle, WA.

Freeman, A.M. 2003. The Measurement of Environmental and Resource Values: Theory and Methods, Second Edition, Resources for the Future, Washington, D.C.

Hulse, D., S. Gregory, and J. Baker. 2002. Willamette River Basin Planning Atlas: Trajectories of Environmental and Ecological Change. Oregon State University, Corvallis, OR.

Lombard, J. 2006. Saving Puget Sound: A Conservation Strategy for the 21st Century. American Fisheries Society and University of Washington Press, Bethesda, MD.

Montgomery, D.R., D.B. Booth, and S. Bolton. 2003. Puget Sound Rivers and Salmon Recovery. In D.B. Montgomery, S. Bolton, D.R. Booth, and L. Wall (eds.), Restoration of Puget Sound Rivers, University of Washington Press, Seattle, WA, pp.1-13.

Montgomery, D.R., S. Bolton, D.B. Booth, and L. Wall (eds.).2003. Restoration of Puget Sound Rivers. University of Washington Press, Seattle, WA.

Mote, P., A. Petersen, S. Reeder, H. Shipman, and L. Whitely-Binder.2008. Sea Level Rise in the Coastal Waters of Washington State. A report by the University of Washington Climate Impacts Group and the University of Washington Department of Oceanography, Seattle, WA.

National Research Council (NRC).2008. Progress Toward Restoring the Everglades: The Second Biennial Review – 2008. National Academies Press, Washington, DC .

Neuman, M., D. St. John, J. Knauer, and N. Salafsky. 2009. Using results chains to develop objectives and performance measures for the 2008 Action Agenda. Puget Sound Partnership, Olympia, WA.

Parametrix. 2003. Chico Watershed Alternatives Analysis: Process and Recommendations. Prepared for the Kitsap County Department of Community Development, Lacey, WA.

Pigou, A.C. 1932. The Economics of Welfare. Fourth Edition. MacMillan, London.

Puget Sound Partnership. 2008. Puget Sound Action Agenda, Protecting and Restoring the Puget Sound Ecosystem by 2020. Olympia.

Ruckelshaus, M.H. and M.M. McClure. 2007. Sound Science: Synthesizing Ecological and Socioeconomic Information about the Puget Sound Ecosystem. Prepared in cooperation with the Sound Science collaborative team. U.S. Dept of Commerce, National Oceanic and Atmospheric Administration (NMFS), Northwest Fisheries Science Center, Seattle, WA.

Shared Strategy for the Puget Sound. 2007. Puget Sound Salmon Recovery Plan. Seattle, WA.

Tietenberg, T. 2006. Environmental and Natural Resource Economics. Seventh Edition. Pearson Addison Wesley. San Francisco, CA

Protection and Restoration Strategies for Watersheds and Tributaries

1. Section Scope

This section reviews, assesses, and summarizes the potential strategies investigated in past scientific and technical research for positively affecting the watersheds and tributaries draining to Puget Sound. The review and assessment covers strategies for both protecting resources that remain and recovering or improving resources that have been impaired. Concentration is on presenting the level of effectiveness of the candidate strategies, as established by the research, and the relative certainty associated with the reported effectiveness. Of particular interest is identifying strategies that reduce multiple threats to the Puget Sound ecosystem.

This section covers the scientific and technical aspects of potential restoration and protection strategies for application in watersheds, whether they drain to tributary fresh waters or to the estuarine and marine waters of Puget Sound. It also encompasses strategies that can be applied within freshwater bodies to affect positively their overall aquatic ecosystems. This scope excludes policy and socioeconomic considerations associated with implementation of the strategies.

A watershed is a unit providing a convenient and practical framework for implementing ecosystem management[1]. Watersheds capture the basic ecological, hydrological, and geomorphological relationships that can be affected by land uses in their drainage catchments (Montgomery, Grant, and Sullivan 1995; National Research Council 1999; Brooks et al. 2003). The concept of a watershed as used in this chapter is a broad one, encompassing the full range of scales that may come into play in describing and evaluating protection and restoration strategies.

Most fundamentally, a watershed is an area of land from which all of the surface and subsurface water drains to a common point. Depending on where the point of interest is located, watershed size can range very widely. The U.S. Geological Survey (USGS) delineates watersheds in the United States using a nationwide system based on surface hydrologic features. This system divides the nation into regions, subregions, accounting units, and cataloging units (National Research Council [NRC] 2009). These hydrologic units are arranged within each other, from the smallest (cataloging units) to the largest (regions). Puget Sound comprises both a subregion and an accounting unit divided into 21 cataloging units ranging in area from 648 to 6634 km^2. The large rivers draining to Puget Sound, or extensive reaches of them, constitute some of these units; others are large tributaries to these rivers; while some are parts of the Sound itself or direct drainages not passing through a major river system. The system put forth by USGS provides a starting point. Ultimately, though, in any particular case a watershed is best defined with reference to specific biogeophysical conditions and problems and management objectives intended to address them. In many cases these considerations will point to a watershed delineation much smaller than a USGS cataloging unit, tens of km^2 down to as small as less than 1 km^2. Of course, relatively small watersheds are nested within larger ones, the components of which are likely to share many characteristics and be amenable to common strategies. This section discusses strategies in relation to their scale of application as appropriate.

Watershed Protection and Restoration within Context of the Puget Sound Partnership's Action Agenda

As outlined in the Introduction to Chapter 4 of the Puget Sound Science Update, the Puget Sound Partnership's (PSP) Action Agenda presented an extensive list of prospective strategies to protect and restore Puget Sound and called out a subset of near-term actions. The technical memorandum Using Results Chains to Develop Objectives and Performance Measures for the 2008 Action Agenda (Neuman et al .2009) incorporated many of these strategies, in some cases supplementing them with others, and grouped them in seven broad categories. The memorandum related strategies to outcomes, threats to the Puget Sound ecosystem, and effects on ecological components, setting up a "results chain" showing how a particular action is thought to lead to some desired result. This section is concerned with the demonstrated effectiveness and relative certainty of some strategies in the Land Protection, Flow Protection, River and Floodplain Restoration, Stormwater, and Wastewater categories.

Organization of this Section

The progression of the section is from general strategies appropriate at relatively large scales, to strategies meant for specific water body types, to management practices for application to particular types of threats. The general strategies apply to scales near or at the USGS cataloging unit level, or even to the entire subregion, and are related primarily to certain strategies in the Land Protection category (Neuman et al. 2009). The section discusses water body-specific strategies for streams (creeks and rivers), wetlands, and lakes. Threat-related strategies are presented for stormwater and wastewater. The stormwater coverage first focuses on urban runoff and is followed by discussions of strategies regarding two other urban issues: municipal and on-site domestic wastewater. The section concludes with a summary of strategies related to runoff from agricultural and forestry production areas.

The identified strategies arise out of the peer-reviewed literature of the subject, which generally requires extensive explanation. Each segment concludes with a synthesis bringing together the threads drawn from the literature. While strategies are usually multi-faceted, an attempt is made at that point to encapsulate the key elements in a relatively brief statement to capture the essence and serve as a touchstone for the strategy.

Terminology

This section has a dual interest in the protection and restoration of the fresh waters tributary to Puget Sound. In the section's usage, "protection" means retaining the ecological state at its existing level, whatever that may be, without diminishment of any indicators of the health of that state, terrestrial or aquatic, structural or functional. The section applies the term "restoration" in the broad sense to mean any level of improvement in the state. This usage is consistent with the definition in Merriam-Webster's Online Dictionary[3], "bringing back to a former position or condition." That definition and the usage in this section carry no connotation of necessarily returning the system to its original state, i.e., pre-human influence, although such an objective is a theoretical possibility. Should such an objective be under consideration, the section refers to the case as "full restoration,' by which it means reestablishment of the structure and function of an ecosystem, including its natural diversity (Cairns 1988, NRC 1992). The section also employs "rehabilitation," when an author has selected that term or for variety of discourse, as a synonym for restoration (Merriam-Webster's defines to rehabilitate as "to restore to a former state").

When referring to lotic (flowing) waters in general, in this section the term "stream" is usually used, which can indicate channels of any size and stream order. When the scale or size is an issue, the section references the order or an actual or approximate range of orders. There are several systems of assigning stream order, but their distinctions are generally not great enough to affect the discussion within the scope of this section. In Strahler's (1952) system, a headwaters stream with no tributaries is first-order, two confluent first-order streams form one of second order, joining with another second-order stream (but not one of first order) becomes a third-order stream, etc.

Framework for Watershed-based Strategies

Background

The National Research Council (NRC) of the National Academy of Sciences convened a committee in 2007, at the request of USEPA, to review its current permitting program for stormwater discharges under the Clean Water Act and provide suggestions for improvement. The broad goals of the study were to understand better the links between stormwater pollutant discharges and ambient water quality, to assess the state of the science of stormwater management, and to make associated policy recommendations. While the committee's charge focused on stormwater and discharge permitting, its report (NRC 2009) offered recommendations more widely applicable to comprehensive water resources management, of which regulatory permitting is of course a key component. This segment of Chapter 3 summarizes the relevant recommendations, in the process building a rationale for a watershed basis and extracting strategies advancing the PSP's Action Agenda and Results Chain process. The approximately 30-year history of stormwater management in the United States has been organized, almost invariably, according to local jurisdictional (city, county) boundaries, with state or USEPA oversight through permitting during the latter half of that period. This organizational principle extends, for the most part, to management of other pollutant-bearing discharges as well. Early in this decade USEPA began to take note of the disadvantages of this practice and the potential benefits of an alternative, in a policy statement (USEPA 2003a) embracing, "... a detailed, integrated, and inclusive watershed planning process ...," with a basis in, "... clear watershed goals ..." Subsequent to the policy statement, USEPA published two guidance documents laying out a general process for setting up Clean Water Act permits on a watershed basis (USEPA 2003b, 2007a). The NRC committee recognized the benefits of and general principles applying to USEPA's concept but concluded that its guidance did not go nearly far enough toward bringing it to fruition. The committee developed an approach fitting within the general framework outlined by USEPA but greatly expanding it in scope and detail. It is intended to replace the present structure, instead of being an adjunct to it, and to be uniformly applied nationwide.

Framework Elements and Their Relationship to PSP Results Chain Strategies

Appendix 4A, Box A1 presents the major elements of effective watershed-based, water resources management and permitting in the committee's approach (NRC 2009), which are elaborated in substantial detail in its report. The list is annotated with identical or similar strategies directly represented in Neuman et al. (2009). These elements represent the following key strategy: *Develop a comprehensive watershed-based management system*.

Comprehensive Watershed Analysis and Management Guidance

A key element in the framework presented above, and in the PSP Action Agenda (PSP 2008), is watershed analysis at an advanced scientific and technical level. Many watershed plans completed over the last 40 years have not been successfully implemented. Davenport (2003), drawing heavily on a survey of practitioners by the Center for Watershed Protection, presented and commented on the main reasons cited for these failures (Box 1).

Box 1. Reasons for failures of past watershed plans (Davenport 2003).

- The plan's scale was too large[2].
- The plan represented a one-time study instead of a long-term commitment.
- The process lacked local ownership.
- The plan skirted real issues about land-use regulation.
- The budget was low or unrealistic.
- Planning focused on the tools of watershed analysis instead of outcomes.
- The document was too long or complex (the backup science and technology should be placed in a separate document).
- The plan failed to assess critically the adequacy of existing programs to implement it.
- Recommendations were too general.
- Regulatory authority for implementation was insufficient.
- Key stakeholders were not involved.
- From the technical standpoint, the plan focused on aggregated averages instead of finer-scale processes, obscuring vulnerable locations and watershed elements that should have been targeted for attention.

Ideas for promising approaches to watershed analysis and management can be found in a number of extended works that have been published over the years explicating multiple aspects of watershed analysis and management. Some of these works are potentially useful to assist the PSP and its collaborators in going forward with the challenging task of strategizing on the watershed level. None is sufficient alone, and none should be used uncritically. Table 1 lists the major, general-purpose works of this type published over the last 15 years and summarizes their characteristics.

Table 1. Characteristics of major publications since 1996 on watershed assessment and management.

		Topics Covered							
Reference	General Features	Processes[a]	Data Collection	Tools[b]	Assessment	Planning	Stormwater Management	Organizing/ Institutional	Financing
Heathcote (2009)	For management professionals, broad scope, interdisciplinary, relatively high level	1-5, 7-9	Minimal	No	Yes	Yes	No	Yes	Yes
USEPA (2008)	For agency staff, qualitative treatment with advice to involve quantitative specialists	1, 4, 7, 8	Yes	1-2, 4, 5	Yes	Yes	Yes	Yes	Yes
DeBarry (2004)	For science, engineering, and planning professionals, broad scope, interdisciplinary, relatively high level	1-5, 7	Yes	1, 2	Yes	No	Yes	Some	No
Brooks et al. (2003)	More forest land than urban emphasis, relatively high level	1-2, 5-6	Brief	1-3 (brief)	Not explicitly	Yes	No	No	No
Davenport (2003)	Planning orientation mainly for regulators and regulated parties	1, 8	No	2, 4	Yes	Yes	No	Yes	No
NRC (1999)	Synoptic coverage on a national scale	1-2	Yes	No	Risk-based	Yes	No	Yes	Yes
Center for Watershed Protection (1998)	Planning process approach at a semi-technical level	No	Some	5-6	Brief	Yes	Brief	Brief	Yes
Reference	General Features	Processes[a]	Data Collection	Tools[b]	Assessment	Planning	Stormwater Management	Organizing/ Institutional	Financing
Terrene Institute (1996)	General coverage at a lay level for decision makers, urban emphasis	1-2	Brief	No	Yes	Yes	Yes	Yes	No

[a] 1—hydrology, 2—water quality, 3—physiography, 4—soils, 5—hydrogeology, 6—sediment transport, 7—ecosystems, 8—land use, 9—social systems

[b] 1—geographic information systems, 2—modeling, 3—statistics, 4—monitoring, 5—mapping, 6—field rapid assessment techniques

It may be seen in Table 1 that no single reference covers all of the principal subjects in detail. However, two relatively recent books, Heathcote (2009) and DeBarry (2004), together do provide quite comprehensive coverage at the level appropriate to the analytical emphasis advocated by NRC (2009). These references would make excellent additions to the library of any scientist, engineer, or planner, or regulator who will be working on Puget Sound issues. Those working on highly forested watersheds should also consider acquiring Brooks et al. (2003). The older material in the table could still benefit those desiring a broader view or who are specialists in the areas stressed by a particular reference. As a manifestation of modern watershed-based, strategy-oriented practice, it is instructive to summarize Heathcote's (2009) general approach. The approach is highly consistent with PSP's strategic emphasis and the NRC (2009) framework (Box 2).

Box 2: General Approach to watershed-based, strategy-oriented management practices proposed by Heathcote (2009).

- Develop an understanding of watershed components and processes and water uses and users.
- Identify or rank problems to be solved or beneficial uses to be protected or restored.

- Set clear and specific goals.
- Develop a set of planning constraints and decision criteria (appropriately weighted).
- Identify an appropriate method of comparing management alternatives.
- Develop a list of management options.
- Eliminate options that are not feasible based on constraints and criteria.
- Test the effectiveness of remaining feasible options using the results of preceding steps.
- Determine the environmental and economic impacts and legal implications of the feasible management options.
- Develop several good management strategies, each encompassing one or more options, for the consideration of decision makers.
- Develop clear and comprehensive implementation procedures for the plan preferred by decision makers.

Four other volumes of a more specialized nature also deserve mention. Field, Heaney, and Pitt (2000) published extended engineering guidance pertaining to urban stormwater management. Its systematic approach has an implicit watershed orientation, although it is not built around watershed assessment and planning in the way that the works in Table 1 generally are. It does cover institutional arrangements and financing, but its treatment of stormwater best management practices (BMPs) is now rather dated, with little explicitly on LID methods. Reimold (1998) compiled material by multiple authors very broad in coverage geographically and by issues. While it gives some attention to analytical and planning tools, this volume is most useful for its regional and problem-area case studies. Pacific Salmon and Their Ecosystems: Status and Future Options, a collection of contributions by many authors edited by Stouder, Bisson, and Naiman (1997), has a fisheries emphasis more suitable for Section 4-4 but is mentioned here because of the watershed-based content in its restoration section. Naiman (1992) edited a volume with the principal title Watershed Management giving many different views of national and even international scope on integrated watershed management and mitigation and restoration.

Strategies for Managing Streams in a Watershed Framework

Research Basis

The condition of streams in urban areas became a subject of study over four decades ago (e.g., Larimore and Smith 1963, Hynes 1970, 1974, Tramer and Rogers 1973, Trautman and Gartman 1974). By the 1990s the effects of watershed urbanization on streams were well documented. They include extensive changes in basin hydrologic regime, channel morphology, and physicochemical water quality associated with modified rainfall-runoff patterns and anthropogenic sources of water pollutants. The cumulative effects of these alterations produce an in-stream habitat considerably different from that in which native fauna evolved. In addition, development pressure has a negative impact on riparian forests and wetlands, which are intimately involved in stream ecosystem functioning (Naiman and Décamp 1997). Much evidence of these effects exists from studies of urban streams in the Puget Sound region and around the United States (e.g., Klein 1979, Richey 1982, Karr, Toth, and Dudley 1985, Pedersen and Perkins 1986, Scott, Steward, and Stober 1986, Garie and McIntosh 1986, Steedman 1988, Booth 1990, Booth 1991, Limburg and Schmidt 1990, Booth and Reinelt 1993, Weaver and Garmen 1994).

Here we report on the findings of two investigations, Booth et al. (2001) and Horner May and Livingston (2003) that address linkages between watershed and aquatic ecosystem elements and the capabilities of prevailing management strategies to influence these relationships. These studies covered primarily second- and third-order streams and their contributing watersheds throughout the region. These streams were chosen because of their instrumental role in supporting both the spawning and rearing life stages of several anadromous salmonid species. Together, the two studies collected data on over 200 reaches on almost 90 streams. Both sought not only to understand the connections among watershed conditions, stream habitat characteristics, and aquatic biology, but also to identify strategies to protect and restore these resources.

Stream Management Strategies Derived from Puget Sound Watershed Research

Both Booth et al. (2001) and Horner,May and Livingston(2003) concluded their work with specific recommendations that supply strategies consistent with the Puget Sound Partnership's Action Agenda and Results Chain strategies. Each study's strategies were highly consistent with one another (see Appendix 4B) and provide the basis for the following key strategy: *Manage stream watersheds using a data- and objective-based approach with appropriate, strategies for streams depending on their levels of ecological condition.*

Table B1 (Appendix 4B) assimilates the recommendations of the contributing studies into a catalogue of strategies set up to meet the intent of the Puget Sound Science Update and fit with the Results Chain memo's structure. Major thrusts of the strategies are reducing the quantities of wet-weather discharges from urban lands and improving the quality of any remaining discharge. Successful implementation of such strategies reduces threats through keeping stream habitats intact, reducing the physical stresses of high flows on stream biota, and decreasing sediment transport resulting from eroded stream channels.

Strategies for Stream Restoration

Introduction

Work to restore the physical habitat and biological communities of streams stretches back decades. This work and its results have been highly documented and the experiences interpreted to formulate extensive guidance on how to perform restoration. Despite this, failures to restore habitats, or even improve them over longer time scales have occurred frequently. Potential reasons for these shortcomings include: the extraordinary complexity of highly dynamic natural systems, the many driving forces operating in these systems not only internally but throughout the climatic and hydrological regimes influencing them, the relative magnitudes and unpredictability of variability in these driving forces and in the biotic and abiotic respondents to them all contribute to the challenges facing restoration efforts. Success in meeting restoration objectives is thus very unlikely if these factors are not appreciated, correctly analyzed, and managed through informed decision making throughout the development of restoration projects.

This segment of Section 3 is concerned with the literature most helpful to understanding the interconnected nature of streams and their contributing watersheds, considerations related to variability, and the techniques available and what they can accomplish.

The prevailing trend in the past has been building in-stream rehabilitation projects to correct localized problems with the objective of improving habitat damaged by altered land use and land cover in the watershed. Isolated problems, such as an improperly sized or configured culvert, are relatively easily identified and corrected. Reversing the consequences of watershed changes, such as channel widening and incision, is a considerably greater challenge, if conditions that led to stream degradation remain unchecked. Nevertheless, many in-stream projects have been constructed in urban or urbanizing basins, attempting to reverse physical and biological degradation in a relatively straightforward and economical manner without addressing the more complex and expensive causes. These projects account for a large share of the failures. All of the references cited here make the point, in one way or another, that the success of any but the most localized in-stream rehabilitation projects is a function of watershed influences. There is no peer-reviewed reference located in the search performed in preparing this chapter claiming otherwise.

The Washington Department of Fish and Wildlife's (WDFW) Stream Habitat Restoration Guidelines (Saldi-Caromile et al. 2004) observed that stream habitat degradation can be caused by: (1) direct physical modification of the stream corridor; (2) changes in channel boundary conditions upstream, downstream, or laterally; (3) physical constraints placed on natural channel adjustment; or (4) changes in watershed management or land uses. Of course, a combination of any or all of those causes might contribute. These categories also broadly delineate sets of techniques that can be brought to bear to rehabilitate habitat. The circumstances that led to ecosystem decline must be identified, followed by developing a set of realistic goals and objectives to reverse or mitigate the decline. Because of limited resources, it is often necessary to prioritize these goals and objectives to target the dominant factors that prevent the reestablishment of the intended ecological conditions (Saldi-Caromile et al. 2004).

Nilsson et al. (2007) addressed the subject of realistic expectations in restoration efforts, stating that it is not self-evident that restoration should try to mimic attributes of previous ecosystems. They pointed out the issue that humans often have priorities contrary to achieving or approaching a pristine ecosystem state, and that human involvement is much more influential now than ever before. Also, the status of previous systems is difficult to establish for several reasons: (1) often, there are no suitable reference systems to mimic, (2) many catchment qualities have changed since the time period chosen for a historic reference system, (3) changes in climate and biota have been continuous in the past, (4) expected climate change is of uncertain magnitude, (5) non-native species cannot be avoided, and (6) landscape context changes through time. To reduce the risk of mistakes, Hughes et al. (2005) recommended that restoration projects should moderate the ambition of identifying specific target states and instead formulate trajectories that accommodate some levels of both variability and unpredictability; i.e., function in an adaptive management framework.

This discussion of stream restoration proceeds, first, to covering key works to guide the conduct of restoration projects, emphasizing those most pertinent to Puget Sound but also covering references of national and international scope. The section then turns to reports on the effectiveness and relative certainty of restoration. Following a passage on climate change implications to stream restoration, it sums up the lessons offered by the literature in relation to the strategies applicable to Puget Sound's streams.

Stream Restoration Guiding Principles: Puget Sound

Extended works on stream restoration tend to be of two types: emphasizing principles and case studies, or offering a practices handbook, guiding the selection and installation of materials and processes to be applied for various purposes. The major references mirror the broad reach and complexity of the subject, described above, and present strategies as diverse as the streams and their watersheds. Hence, exposition of detailed aspects of their coverage and highly specific strategy discussions are beyond the scope of this review. Instead, our review concentrates on the general features and areas of coverage from past studies and focuses how they can be used by Puget Sound practitioners.

Guidance Emphasizing Principles and Case Studies

Montgomery et al. (2003) supplies strong intellectual underpinning for assessing, planning, and designing restoration projects for Puget Sound's streams. The volume provides a highly systematic and well grounded treatment of the subject by multiple authors, proceeding from geological and geomorphological controls on stream characteristics and dynamics; to aquatic biological aspects; onward to chapters addressing social constraints on action; and then to the application of concepts from fluvial geomorphology, civil engineering, riparian ecology, and aquatic ecology to restoration projects. While not comprehensive on structural tools, Montgomery et al. (2003) does have three chapters on various aspects of using wood, a key element of Puget Sound streams often missing or greatly reduced in degraded systems in the region. Rather, it concentrates more on the criteria that must be applied and why (e.g., the complete and prioritized design criteria for an actual project presented by Miller and Skidmore (2003). The book concludes by raising five large questions (Box 3) that the authors recommend be kept in mind to guide restoration, the answers to which they believe will determine the likelihood of success (Bolton, Booth, and Montgomery 2003). The reader is directed to locations within the text to find guidance in formulating answers:

Box 3. Five large questions to guide restoration of rivers and streams in the Puget Sound (from Montgomery et al. 2003)

1. What is the physical template upon which restoration will take place?
2. Is the watershed urbanized, agricultural, or forested?
3. Is the river being restored large or small?
4. Is there a thorough watershed assessment that identifies historic and current habitat-forming processes and fish distribution?
5. Has a monitoring plan been developed in concert with the planned restoration action?

Wissmar and Bisson (2003) provided another, more specialized reference of substantial potential use to Puget Sound-region stream efforts. Composed of 12 papers by mostly regional authors, this book's concern is variability and uncertainty in applying stream restoration strategies. With effective restoration of aquatic habitat depending upon reestablishing watershed and stream processes to a range of variability that maintains dynamic equilibrium (Saldi-Caromile et al. 2004), these considerations are obviously crucial to success, and are informed in theoretical and applied models by the papers collected by Wissmar and Bisson (2003).

Additionally, Darby and Sear (2008) took up the theme of Wissmar and Bisson in 14 more papers on the subject. Most papers include case studies from Europe and North America.

Guidance Emphasizing Practices

WDFW's Stream Habitat Restoration Guidelines (Saldi-Caromile et al. 2004) begin with a chapter covering watershed processes and stream and floodplain processes and attributes. It lays out some important fundamentals in approaching stream restoration, which are paraphrased here:

Box 4. Fundamentals in approaching stream restoration from Saldi-Caromile (2004)

Channel and floodplain structure, and aquatic habitat, are created, maintained, and destroyed by the energy inherent in high flows. Complex patterns of the resulting sediment erosion and deposition underlie diverse, productive aquatic and riparian habitat. A stream reach in dynamic equilibrium has developed a geometry that balances the energy available for sediment transport with the supply of sediment being delivered to the reach. With this balance, individual channel and floodplain features are created and destroyed but overall channel characteristics such as sinuosity, gradient, width/depth relationships, and pool and riffle frequency are maintained. The stabilizing role of vegetation in channel development and maintenance cannot be overemphasized. Channel complexity, having a large effect on energy dissipation, exerts a major influence on erosion, sediment transport, and deposition, and hence on dynamic equilibrium.
A stream reach undergoing simplification of overall channel characteristics falls into disequilibrium. If the disturbances are temporary, the stream will often recover its former characteristics. Chronic alterations to watershed and stream processes exceeding the natural range of variability of those processes will inevitably alter the stream habitat and ecosystem, eventually to a new, often simplified, equilibrium state. Effective restoration of aquatic habitat depends upon reestablishing watershed and stream processes to a range of variability that maintains a complex channel/floodplain system in dynamic equilibrium.

The lesson for one who would restore a stream, then, is to understand the habitat requirements of the target biota, understand the tolerances of variability to maintain those habitat attributes, and create conditions remaining within those ranges of tolerance. Gaining this understanding implies a need to perform careful watershed, floodplain, and channel assessments as a prerequisite for restoration planning and design. Unfortunately, the WDFW's assessment chapter remains in a draft, incomplete form. However, stream assessment protocols are numerous; Somerville and Pruitt (2004), contracted by USEPA, reviewed 45 protocols developed for national or regional application. They concluded that a protocol should have the followingcharacteristics:

- *Classification*: Stream assessment should be preceded by classification to narrow the natural variability of physical stream variables.
- *Objectivity*: The assessment procedure should remove as much observer bias as possible by providing well-defined procedures for objective measures of explicitly defined stream variables.
- *Quantitative methods*: The assessment procedure should utilize quantitative measures of stream variables to the maximum extent practicable.

- *Fluvial geomorphological emphasis*: Stream assessments undertaken to prioritize watersheds or stream reaches for management or aid the design of stream enhancement or restoration projects should be based on fluvial geomorphic principles.
- *Data management*: Data from stream assessments should be catalogued by designated entities in each region of the country.

There has been no Puget Sound regional agreement on a full assessment protocol to support stream restoration. The advice of the USEPA-sponsored review should be taken to settle on the best instrument for the region's purposes and insert it in the WDFW guidance.

The WDFW manual (Saldi-Caromile et al. 2004) is much more complete in presenting restoration strategies compared to its coverage of assessment. It guides developing and prioritizing goals, objectives, and activities for habitat preservation, restoring habitat-forming processes and connectivity, and modifying and creating stream habitats. It then goes on to elaborate factors to consider when identifying and selecting ecosystem recovery alternatives. Specific approaches are given for restoring sediment supply, the flow regime, energy inputs, water quality, incised and aggrading channels, and salmonid spawning and rearing habitat. The final chapter, the appendices, and a separate series of white papers cover the many techniques, in 15 categories, for achieving restoration goals and objectives. Presentation of each technique includes a description, potential positive and negative physical and biological effects, appropriate applications, risk and uncertainty, design guidance, permitting, construction considerations, costs, monitoring, maintenance, and examples. It also offers general cautionary and cogent advice relevant to any restoration undertaking.

Allied with the WDFW Stream Habitat Restoration Guidelines (Saldi-Caromile et al. 2004) is The Integrated Streambank Protection Guidelines, also from WDFW (WDFW 2003a). These guidelines use a series of sequential or hierarchic matrices to aid in selection of practices applying to the specific restoration task of treating eroding stream banks.

National and International Guidance

The Natural Resources Conservation Service's (NRCS, 2007a) *Stream Restoration Design, National Engineering Handbook, Part 654* is a key reference on the national scale. The handbook is very comprehensive and detailed; but, as its authors regularly caution, the approaches and techniques are not necessarily appropriate in all circumstances. That caution should be particularly taken under consideration in working with Puget Sound's unique salmonid streams.

Table 2 summarizes the areas of coverage of major works published over the last 15 years devoted mainly to stream restoration principles and general strategies and case studies. All have multiple authorships. The *Federal Interagency Stream Restoration Working Group* (1998) manual includes coverage of stream assessment and specific techniques for in-stream application, although not in the detail of the NRCS handbook.

Table 2. Topics of major publications since 1996 covering stream restoration principles, general strategies, and case studies

Reference	Restoration Principles	Stream Processes	General Strategies	Organizing/ Institutional	Case Study Scope
Brierley and Fryirs (2008)	Yes	Yes	Yes	No	World
de Waal, Large, and Wade (1998)	Yes	No	Yes	No	World
Federal Interagency Stream Restoration Working Group (1998)	Yes	Yes	Yes	Yes	None fully developed
Williams, Wood, and Dombeck (1997)	Yes	No	Yes	Yes	United States
Brookes and Shields (1996)	Yes	Yes	No	No	United States, Europe

Effectiveness and Relative Certainty of Stream Restoration

Assessment of In-Stream Habitat Restoration Projects

The tendency throughout most of the history of stream restoration has been to place structures intended to rehabilitate habitat. As a result, most effectiveness evaluations found in the literature concern these practices. Frissell (1997) compiled the results of seven studies of restoration project failures or unanticipated outcomes performed between 1956 and 1992. These projects involved placement of deflectors, check dams, gabions, and/or cover structures in western United States streams draining watersheds disturbed by grazing, mining, or logging without upland mitigation. Majorities of the emplaced structures were damaged or destroyed in every one of the six cases registering physical conditions within 1 to 18 years after installation. Two studies recorded biological (fish or frog) residence, which declined from pre-restoration in both cases. The author attributed the poor results on failure to appreciate the difference between "strategy," by which he meant comprehensive, large-scale, long-term actions, and "tactics," which he defined to be a localized, short-term approach.

Brown (2000) examined 22 different types of in-stream restoration practices, involved in more than 450 total installations, classified in four categories: (1) bank protection, (2) grade control, (3) flow deflection or concentration, and (4) bank stabilization. He evaluated each in the field according to the visual criteria structural integrity, function, habitat enhancement, and vegetative stability. He found about 90 percent the practice types to be appropriate for use in the applications investigated, and that around the same proportion of the individual installations remained intact after an average of four years. Unintended scouring or sediment deposition occurred in 20-30 percent of the cases, and less than 60 percent fully achieved habitat enhancement objectives. Most failures were associated with projects that attempted to create different channel plan-form geometry, generally a pre-disturbance channel morphology type superimposed on a stream in a disturbed watershed.

Jacobsen et al. (2007) assessed 163 in-stream and riparian projects completed by the Western Oregon Stream Restoration Program on 285 km (178 miles) of stream during 2002-2004 to boost coho salmon and/or steelhead production. The majority of projects were large wood placements (116), followed by stream fencing (20), fish passage (15), riparian planting (seven) and boulder placement (two). The assessments documented physical changes expected to be beneficial to stream function and productivity of salmonids (there was no direct measurements of fish presence). Most surveys were conducted within one year following treatment. The restoration activities were effective at improving overall habitat complexity and ecological conditions, although a significant increase in quality of over-wintering habitat for juvenile coho salmon was not observed. Fish habitat models (Habitat Limiting Factors Model Version 7.0 and HabRate Version 2.0) did not demonstrate a large increase in habitat quality, apparently at least in part because the amount of off-channel and slow-water pool habitat did not increase significantly, although the treatments increased the complexity of pool habitat.

Installation of large woody debris (LWD) or boulders has become one of the most common techniques to improve fish habitat and compensate for its simplification. LWD, defined[4] as a loghaving a mid-point diameter of at least 10 cm, a length of 2 meters, and protruding into the bank-full channel (although sometimes with variations in those dimensions), is particularly important in Pacific Northwest streams. It provides roughness that regulates velocity, cover to fish, and aid in forming pools in which fish feed and rest (McMahon and Hartman 1989). LWD also influences bank stability, sediment retention, and channel grade (Montgomery et al. 1995, Beechie and Sibley 1997, Nelson 1999).

Booth et al. (2001) investigated restoration efforts at six streams, determining the response of invertebrates to LWD placement. The six projects lay in physically similar watersheds but with widely different levels of human disturbance. No anchored LWD moved at any of the project reaches, and where over half of the unanchored logs were key pieces (individual logs with attached root wads) there was also no substantial LWD movement. In the projects with unanchored LWD and few or no key pieces, however, LWD movement was documented. Both types of LWD addition raised pool numbers, at least slightly, towards those of less disturbed streams; but post-project pool spacing was not correlated to LWD loading. Addition of LWD had little demonstrable effect on biological condition as measured by Benthic-Index of Biological Integrity (B-IBI). B-IBI did not increase either when sampling sites were located within or downstream of project boundaries. These results indicate that, although projects several hundreds of meters long improved an important measure of physical habitat (pool spacing) in a stream reach over a time scales of 2-10 years, they had little influence on the benthic invertebrate assemblage.

The effectiveness reports cited thus far lack evaluation of restoration project success in increasing fish productivity. James (2007) attempted to make this connection for Washington State projects having the objective of salmonid population increases. He found that fish passage projects increased adult coho salmon relative abundance and juvenile salmonid densities, but the increases were not statistically significant. In-stream habitat improvement projects significantly increased residual pool metrics, LWD volume, and water surface area. Multiple regression analyses showed relationships between habitat features and fish species distribution, but the author did not report on population changes of the target species before and after restoration projects. James' (2007) major conclusion was that demonstration of biological outcomes requires monitoring involving more spatial and temporal replication and detailed data using a suite of metrics.

Assessment of a Broad Range of Stream Restoration Techniques

Roni et al. (2002) presented an influential paper that reviewed a wide range of stream restoration techniques and proposed a hierarchical strategy for prioritizing restoration in Pacific Northwest rural watersheds moderately modified by activities like logging and rangeland practices. They placed the techniques in six broad categories: (1) habitat reconnection (e.g., culvert improvements, off-channel connections), (2) road work (e.g., removal, improvement), (3) riparian vegetation restoration, (4) in-stream habitat restoration (e.g., wood and boulder placement), (5) nutrient enhancement, and (6) habitat creation (e.g., in-stream with wood and boulders, off-channel).

In their review, Roni et al. (2002) found that reconnecting isolated off-channel habitats or blocked tributaries provides a quick biological response, is likely to last many decades, and has a high likelihood of success. They recommended that these types of restoration activities be undertaken before methods that produce less consistent results. Riparian restoration or road improvement may not produce results for many years or even decades. Roni et al. (2002) reviewed eight studies documenting LWD or boulder persistence in a functioning state. Two of these studies were also reviewed by Frissell (1997), but seven of the eight in the review of Roni et al. (2002) were more recent, from the 1990s. These authors found a higher rate of success by that point in time, with 85 percent of artificially placed wood remaining in place and contributing to habitat formation. They attributed the improvement to an increased emphasis on replicating natural architecture of wood in streams (creating natural jams or pinning logs between riparian trees), instead of employing artificial structures (e.g., weirs, deflectors), although they noted that the supporting studies were of short duration. The available evidence still suggests that most in-stream structures persist for less than 20 years.

Roni et al. (2002) cite 11 papers summarizing biological evaluations of 29 in-stream restoration projects for anadromous fish in the Pacific Northwest. Post-treatment juvenile abundance for at least one species or life stage increased significantly in 12 streams or was higher in treatment reaches than in control reaches. However, in only five of these studies (six streams) were populations monitored for more than 5 years.

In summarizing their review of the literature on in-stream habitat restoration techniques, Roni et al. (2002) concluded that LWD projects are effective at creating juvenile coho salmon rearing habitat and increasing juvenile densities, but the response of other species is less clear. Although increased spawner densities have been reported in some studies, there are no thorough evaluations of the response of spawning adults to structure placement. Artificial structures such as log weirs and deflectors appear to have moderate-to-high failure rates, and their benefits to fish may be temporary. Therefore, placement of LWD and other material in the stream channel should mimic natural processes by using and placing materials consistent in size, type, location, and orientation to that found in natural channels.

Roni et al. (2002) presented a flow chart depicting a hierarchical strategy for prioritizing specific restoration activities. They advocated implementing first those techniques that have a high probability of success, low variability among projects, and relatively quick response time. As noted above, habitat reconnection was found to meet those criteria best. Riparian restoration and road improvement should be considered after reconnecting high quality, isolated habitats. In-stream LWD and other structural placements should either be undertaken after or in conjunction with reconnection of isolated habitats and efforts to restore watershed processes. The authors noted that their framework may need modification for use in highly altered agricultural and urban watersheds, where some processes cannot be reliably restored or where water quality and hydrologic changes may compromise the effectiveness of many of the commonly employed restoration techniques.

Roni et al. (2002) also note that, while they focused on restoration, it is important not to overlook the need to protect high-quality habitats. This protection should be given priority over habitat restoration, because it is far easier and more successful to maintain good habitat than to recreate or restore degraded habitat.

The reports on in-stream structural projects to improve habitat discussed earlier do not strongly encourage optimism for effective restoration with much certainty. However, the work of Roni et al. (2002) gives a brighter picture, at least for restoring streams not affected by highly altered watershed conditions. The authors' hierarchical strategy places the major categories of stream restoration activities in context and proper order. They sensibly recommend withholding the prevailing technique of in-stream structural placement at least until habitat reconnection is accomplished, and then implementing that technique by mimicking natural materials and processes. Their advice on giving priority to protection over restoration is particularly pertinent to streams in areas subject to urbanization, which they believe are at a considerable disadvantage for restoration compared to the mostly rural streams in their database.

Further Quantification of Restoration Effectiveness and Relative Certainty

Recently published work further develops the assessment of restoration effectiveness and relative certainty. Two studies (Palmer et al. 2010 and Miller et al. 2010) used statistical and numerical analysis techniques to examine broadly biological responses to restoration actions intended to recover habitat heterogeneity. A third contribution (Stewart-Koster et al. 2010) presented a quantitative decision-making tool to select among restoration alternatives in an environment subject to multiple driving forces.

Palmer et al. (2010) evaluated studies that quantitatively examined the reach-scale response of invertebrate species richness to restoration actions that increased channel complexity and habitat heterogeneity. Adopting these objectives to regain biodiversity has become a dominant paradigm in ecological restoration. This paradigm is reflected in stream restoration projects through the common practice of re-configuring channels to add meanders and adding physical structures such as boulders and artificial riffles. They also evaluated studies that used manipulative or correlative approaches to test for a relationship between physical heterogeneity and invertebrate diversity in streams that were not in need of restoration.They also used habitat and macroinvertebrate taxa richness data from 78 independent stream restoration projects described by 18 different author groups. Most projects were successful in enhancing physical habitat heterogeneity; however, only two showed statistically significant increases in biodiversity rendering them more similar to reference reaches or sites. Studies manipulating structural complexity in otherwise healthy streams were generally small in scale, and less than half showed a significant positive relationship with invertebrate diversity. Only one-third of the studies that attempted to correlate biodiversity to existing levels of in-stream heterogeneity found a positive relationship. Across all the studies evaluated, there is no evidence that habitat heterogeneity was the primary factor controlling stream invertebrate diversity, particularly in a restoration context. The findings indicate that physical heterogeneity should not be the driving force in selecting restoration approaches for most degraded waterways (Palmer et al. 2010). Evidence suggests that much more must be done to restore streams impacted by multiple stressors than simply re-configuring channels and enhancing structural complexity with meanders, boulders, wood, or other structures (Palmer et al. 2010).

Palmer et al. (2010) concluded by observing that, as integrators of all activities on the land, streams are sensitive to a host of stressors, including, depending on the watershed, impacts from urbanization, agriculture, deforestation, invasive species, flow regulation, water extractions, and mining. The impacts of these factors individually or in combination typically lead to a decrease in biodiversity because of reduced water quality, biologically unsuitable flow regimes, dispersal barriers, altered inputs of organic matter or sunlight, degraded habitat, etc. Despite the complexity of these stressors, a large number of stream restoration projects focus primarily on physical channel characteristics. They asserted that this practice is not a wise investment if ecological recovery is the goal and that managers should critically diagnose the stressors impacting an impaired stream and invest resources first in repairing those problems most likely to limit restoration (Palmer et al. 2010). It might be added to the authors' conclusions that if correcting such problems would require more resources than available, then protection or restoration activities should be pursued elsewhere, where the resources that can be marshaled can make a positive impact with more certainty.

Miller et al. (2010) employed meta-analysis to examine the relationship between restoring physical habitat heterogene¬ity and macroinvertebrate response in terms of diversity and density. Meta-analysis compares results among studies through computation of a common-size effect, scaled by unit variance and weighted by sample size. They analyzed monitoring results from 12 replicated and 14 un-replicated projects, the majority intended to reverse or mitigate the effects of stream channelization. The researchers determined that increasing habitat heterogeneity had significant, positive effects on macroinvertebrate richness. Macroinvertebrate density also increased, although not statistically significantly (Miller et al. 2010). Large woody debris additions produced the largest and most consistent responses, whereas responses to boulder additions and channel reconfigurations were positive, yet highly vari¬able. For example, in the replicated studies richness and density increases were, respectively, 83 and 75 percent greater on average for LWD compared to boulder additions. Among all strategies, the strength and consistency of macroinvertebrate responses were related to land use orwatershed-scale conditions, but appeared independent of project size, stream size, or recovery time.

Stewart-Koster et al. (2010) proposed and demonstrated a solution to the problem of making restoration decisions in the face of multiple system alterations and stressors (e.g., changes in flows, catchment and riparian land-use, habitat degradation, modification of stream energy regimes). Their solution predicts not only the effectiveness of alternative actions but also the relative certainty associated with them. They employed Bayesian networks as a decision support tool for considering the influence of multiple factors on aquatic ecosystems and the relative benefits and costs of various restoration options.

A Bayesian network is a probabilistic graphical model representing a set of factors of a system (nodes) as random variables and their conditional dependencies. The dependencies are depicted as directed links connecting a "parent node" to a "child node." The network is quantified by populating conditional probability tables associated with the nodes in the network. The table entries can be specified by experts or derived from data. Efficient algorithms exist to draw inferences and perform decision analysis in Bayesian networks. Stewart-Koster et al. (2010) also modified the Bayesian networks to incorporate the relative costs and benefits of potential management actions. Such models are known as Bayesian decision networks and are used interactively to identify the most appropriate decision (here, restoration action) given estimated costs and benefits. A key advantage of Bayesian techniques is that they can easily be operated to give the relative certainty of the predictions.

Reconnecting Habitat by Removing Blockages to Fish Passage

Roni et al. (2002), with some subsequent support by James (2007), established that habitat reconnection projects in general, and fish passage improvement works in particular, are the most effective and certain restoration strategies to improve fish production. Roni et al.(2002) recommended them for first priority among the suite of strategies. Because of their primacy, this segment of the chapter gives brief attention to some specifics of their implementation.

Roni et al. (2002) observed that, among the alternatives for removing blockages, bridges generally allow the free passage of materials and formation of a natural stream channel but are costly. Open-bottom culverts or embedded (e.g., countersunk) pipe-arch culverts allow a natural substrate to form within the channel and are effective at passing both juvenile and adult salmonids (citing Furniss et al. 1991 and Clay 1995). However, such culverts can constrain the stream channel, if the culvert size does not account for large flows or the volume of sediment and wood transported by the stream (citing Robison 1999). Other design options include backwatering culverts at the outlet or inlet and placing baffles within the culvert to reduce flow velocity.

Roni et al. (2002) gave a useful summary of various stream crossing structures and whether or not they allow for juvenile and adult salmonids fish passage and the transport of sediment and LWD or impact stream morphology by constraining the channel (Table 3). The advantage of bridges shows clearly in the table. Otherwise, the bottomless pipe arch and squash pipe or countersunk culvert types supply the most passage benefits, although they still constrain the channel.

Table 3. Summaryof Fish Passage and Material Transport Capabilities and Channel Constraints of Various Stream Crossing Structures, after Roni et al. (2002).

	Provides FishPassage for:		Transports:		
Stream Crossing Type	Adults	Juveniles	Sediment	LWD	Constrains Channel[a]
Bridge	Yes	Yes	Yes	Yes	No
Culverts:					
Bottomless pipe arch	Yes	Yes	Yes	No	Yes
Squash pipe or countersunk	Yes	Yes	Yes	No	Yes
Round corrugated, baffled	Yes	Yes	No	No	Yes
Round corrugated, no baffles	Yes or No[b]	Yes or No[b]	No	No	Yes
Smooth (round or box)	No	No	No	No	Yes

[a] The degree of constraint depends upon the size of the culvert or bridge relative to the channel and floodplain width.

[b] Fish passage depends upon culvert slope and length.

The references cited below can be used by qualified scientific and technical personnel to work through an entire fish passage project development, from collecting the needed data, to analyzing and selecting among options, through design, and on to construction and long-term maintenance.

Clay (1995) reviewed fish passage and the techniques to effectuate it under a broad range of circumstances, including at road crossings. He provides an initial primer on culvert designs and methods for retrofitting impassible culverts, although not in sufficient detail for design.

WDFW's *Design of Road Culverts for Fish Passage* (WDFW 2003b)serves as guide for designing permanent road-crossing culverts to facilitate upstream fish migration (the manual does not explicitly cover downstream migration). It provides guidance for projects involving new culvert construction as well as retrofitting or replacing existing culverts, laying out the consecutive design steps most likely to be required in a culvert project. The guide emphasizes, as a first step, determining if a culvert is a suitable solution for providing fish passage at the particular site in question. Wherever a roadway crosses a stream, it creates some level of risk to fish passage, water quality or specific aquatic or riparian habitats. Any and all alternatives should be investigated to minimize the number of sites where a roadway crosses a stream, including designing road alignments to avoid crossings and consolidating crossings, with bridges preferred where crossings must occur.

The WDFW handbook (WDFW 2003b) recognizes three design options: (1) No-Slope Design Option, (2) Hydraulic Design Option, and (3) Stream-Simulation Design Option. The No-Slope Design Option results in reasonably sized culverts without requiring much calculation. The Hydraulic Design Option requires hydrologic and open-channel hydraulic calculations, but it usually results in smaller culverts being required than the No-Slope Design Option (smaller culverts may trap more debris, however, and a factor of safety must be applied). The Hydraulic Design Option is based on velocity, depth and maximum turbulence requirements for a target fish species and age classes. The Stream-Simulation Design Option involves constructing an artificial stream channel inside the culvert, thereby providing passage for any fish that would be migrating through the reach. It is difficult in most situations, if not impossible, to comply with velocity criteria for juvenile fish passage using the Hydraulic Design Option. The No-Slope and Stream-Simulation Design options, on the other hand, are assumed to be satisfactory for adult and juvenile passage; thus, they tend to be used more frequently at sites where juvenile fish passage is required. Application of the No-Slope Design Option was determined to be most effective for relatively short culverts at low-gradient sites (WDFW 2003b)

Frei (2006) produced a university thesis that became Federal Highway Administration Hydraulic Engineering Circular (HEC) – 26, a design reference for the classification, design, and installation or retrofit of a stream crossing (culvert or bridge) ensuring fishpassage. The document assumes no particular set of passage criteria; rather, it compiled design options endorsed in different geographic regions to allow the user to select the most appropriate design method for their situation. A collection of design examples and case histories illustrates the design methodology selection. The manual follows a logical progression to guide the reader through the assessment and design process. Culvert and bridge analysis, design, and retrofit techniques are then described, followed by case histories and design examples. Specific requirements of the fish species in question and the hydrologic and geomorphologic circumstances demand that the design for fish passage be considered on a site-by-site basis, all but eliminating the possibility of a “cookie-cutter” design approach. The author noted the diverse expertise needed for a stream crossing effective for fish passage, generally including stream ecology; fish biology; hydrology; and hydraulic, structural, and geotechnical engineering (Frei 2006).

The Federal Highway Administration (FHWA, 2007) issued a synthesis report covering design for fish passage at stream crossings. Extending from HEC – 26, it places culvert design techniques into four categories based on design premise and objectives: (1) No-Impedance techniques, which span the entire stream channel and floodplain; (2) Geomorphic-Simulation techniques, which create fish passage by matching natural channel conditions within the culvert crossing; (3) Hydraulic-Simulation techniques, which attempt to resemble hydraulic diversity found in natural channels through the use of natural and oversized substrate; and (4) Hydraulic-Design techniques, which may utilize roughness elements such as baffles and weirs to meet species-specific fish passage criteria. Preliminary chapters covering the topics of fish biology and capabilities, culverts as barriers, fish passage hydrology, and design considerations aid in the selection of appropriate design techniques based on hydraulic, biological, and geomorphic considerations. A further section presents examples of design techniques fitting the defined design categories. Design examples and case histories for a selection of design techniques are presented next, and are followed by a discussion on construction, maintenance, monitoring, and future research needs. The FHWA synthesis (FHWA 2007) provides comprehensive, highly quantitative guidance and amply represents Washington State methods and examples.

Climate Change and Stream Restoration

The Climate Impacts Group (CIG 2009) used Intergovernmental Panel on Climate Change (IPCC) data predicting Pacific Northwest average temperature and precipitation increases (covered in Chapter 3 of the Puget Sound Science Update) to forecast effects on Washington State hydrology and water resources (CIG 2009). Projections show seasonal river flow timing shifting substantially in snowmelt-dominated and rain-snow mixed watersheds. Although anticipated overall increases are relatively small, a shift is expected to more precipitation in the cooler, wetter season and less in the warmer, drier season. This shift translates to proportionately greater runoff increases, because of the higher efficiency of cool-season precipitation in producing runoff, a phenomenon associated with the lower available soil moisture storage capacity and vegetative demand for water in the cooler season.

The CIG (2009) runoff predictions were applied to estimate stream flow for four snow-rain mixed Puget Sound rivers (the Cedar, Sultan, Tolt, and Green rivers). Stream flow is a quantity related to but differing from runoff, because of the time-lag effect associated with the intervening hydrologic processes. Modeling showed a consistent shift in the hydrographs of all four basins toward higher cool-season and lower warm-season discharges (CIG 2009)

The aforementioned temperature and hydrologic predictions were applied to evaluate the sensitivity of Pacific salmon (*Oncorhynchus spp.*) (Mantua et al. 2009). It is expected that the combined effects of warming stream temperatures and altered flows will reduce the reproductive success of many salmon populations, but vary according to life histories and watershed characteristics. Populations with extended freshwater rearing periods (steelhead [*Oncorhynchus mykiss*], coho [*Oncorhynchus kisutch*], sockeye [*Oncorhynchus nerka*], and summer Chinook [*Oncorhynchus tshawytscha*]) were forecast to experience large increases in summer-time thermal and hydrologic stresses (Mantua et al. 2009). Other populations with brief freshwater rearing periods were projected to exhibit the greatest productivity declines in snow-rain mixed rivers, where anticipated increased winter flood magnitudes and frequencies will reduce egg-to-fry survival rates. Summer chum salmon (*Oncorhynchus keta*) are especially vulnerable because of their reliance on small, shallow streams in the late summer and early fall (Mantua et al. 2009). The Lake Washington Ship Canal is already thermally impaired and inhibiting to certain adult and juvenile salmon migrations, a condition expected to be aggravated in the future.

Mantua et al. (2009) note that many of the hydrologic processes highly sensitive to climate change are also known to be similarly sensitive to land and water use impacts. They went on to recommend as strategies to mitigate stream temperature increases:

1. Reducing water withdrawals in warm, low-flow periods

2. Restoring floodplain functions that recharge aquifers
3. Protecting and restoring thermal refugia provided by groundwater and tributary inflows, undercut banks, and deep pools
4. Restoring riparian shade
5. Protecting and enhancing summer in-stream flows (e.g., by strategic reservoir releases)

Their recommended strategies to reduce risks to salmon by elevated fall and winter flows included:

1. Protection and restoration of off-channel habitat in floodplains as refugia
2. Limiting expansion of effective impervious area
3. Retaining watershed forest cover
4. Operating reservoirs to reduce flooding.

Ormerod (2009) put forth similar recommendations in considering river management on the world-wide scale in the climate-change era, emphasizing increasing landscape-scale connectivity, reducing population vulnerability to other negative effects, and strengthening protected-area networks.

Battin et al. (2007), preceding the work by CIG (2009), used a series of linked models of climate, land cover, hydrology, and salmon population dynamics to investigate the impacts of climate change on the effectiveness of proposed habitat restoration efforts designed to recover depleted Chinook salmon populations in the Snohomish River basin. Their results were in accord with the subsequent work in predicting thermal impacts and hydrologic shifts negatively affecting salmon spawning and incubation, a large negative impact on freshwater salmon habitat, and the particular vulnerability of snow-rain mixed rivers. With the most pronounced effects expected to occur in high-elevation streams with a generally high level of protection through federal ownership and little restoration potential, restoration efforts will be mounted at lower elevations. The authors expect that the combination of extensive impacts high in watersheds and restoration concentrated at lower altitudes will cause a spatial shift in salmon abundance (Battin et al. 2007).

Battin et al. (2007) also used their suite of models to examine the interaction between climate andrestoration effects for three future (i.e., 2025) land-use scenarios: a scenario representing no change from "current" (2001) conditions, one based on a linear future projection of current land-use change and population trends that includes the completion of current restoration projects but no further restoration ("moderate restoration"), and a scenario in which all restoration targets in the restoration plan are met ("full restoration"). The moderate restoration scenario resulted in slightly higher minimum spawning flows and incubation peak flows but lower pre-spawning temperatures than the current scenario. The full restoration scenario resulted in somewhat lower incubation peak flows, with little change in pre-spawning temperatures. Spawning flows decreased slightly under the full restoration scenario because of greater evapotranspiration from the increased forest cover. The authors concluded that although expected climate impacts cannot be mitigated entirely, habitat restoration can play an important role in offsetting those effects.

Palmer et al. (2009) wrote that the anticipation of climate change impacts requires a proactive management response if valuable river assets are to be protected. Furthermore, a proactive response requires sound monitoring and predicting capabilities at the scales that management actions can be applied, which, they believe, is almost always at the local watershedscale (Palmer et al. 2009). They recommend such an approach because of evidence that factors such as urbanization will interact with climate change in ways that are likely to determine the impacts to aquatic biota (Nelson et al. 2009). Increased efforts at both protection and restoration are major components of the program they recommend. In lower-elevation areas, they contend that increasing protection of and restoring floodplains and riparian corridors will not only provide protection for river ecosystems but also will reduce the impacts of both floods and droughts on humans. Giving more watersheds protected status, particularly those at higher elevations expected to experience the most dramatic climate changes, can thus provide refuge habitat for species under multiple threats (Palmer et al. 2009).

Synthesis of Stream Restoration Strategies

In-stream restoration strategies summarized here directly address multiple threats to Puget Sound's tributary streams, including restriction of anadromous fish passage, salmon spawning and rearing habitat degradation, and stream food web disruption. To the extent that watershed restoration accompanies in-stream rehabilitation, strategies address the additional threats of stream channel modification; acute and chronic toxicity effects on aquatic organisms from metal and organic pollutants; and increased pollutant loadings to all downstream waters, including Puget Sound.

The strategies are drawn from the literature reviewed above and are framed in a process of assessment of problems and conditions, development of restoration concepts, and design of restoration elements. In general, problems fit into the categories of physical modification of the stream corridor, changes in channel boundary conditions, physical constraints on channel adjustment, watershed changes in management, or a combination (see Saldi-Caromile et al. 2004). Montgomery et al. (2003) can be consulted to guide assessment of the type and extent of problems, and then to move toward concept development. The forecasts of the Climate Impacts Group (2009) and future work by that group can inform future assessment and concept development.

The literature exhibited a strong consensus that, before restoration proceeds, consideration be given to protecting well functioning streams and their habitats as well as necessary actions in the contributing watershed to achieve restoration goals and objectives. A corollary is that if these actions are infeasible for any reason and cannot be performed, that goals and objectives be adjusted to what is attainable without mitigation of watershed-based problems.

A key component of overall watershed restoration is rehabilitation of the riparian zone, which as shown by the research reported above, must be relatively wide, continuous, and covered by mature vegetation to advance effective restoration of the adjacent stream. Once possible in-stream options are identified, the hierarchical strategy of Roni et al. (2002) is a mechanism to prioritize among them in relation to assessment results. That strategy emphasizes habitat reconnection as generally the most effective and certain of in-stream strategies, where prior disconnection is among the problems. The strategy then guides a user through consideration of riparian restoration and road improvements, with in-stream structural placements to follow or occur simultaneously with any of the other actions, as appropriate.

In complex cases involving multiple stressors, including climate change, the Bayesian approach of Stewart-Kloster et al. (2009) holds promise as a means of objectively assessing effectiveness and relative certainty of alternative actions. Papers in Wissmar and Bisson (2003) and Darby and Sear (2008) can also assist in grappling with variability and uncertainty.

Key Strategy: Synthesis of guiding principles for stream restoration

- Protect well functioning streams and their habitats, where they exist.
- Consider which actions are necessary in the contributing watershed to achieve restoration goals and objectives.
- Identify in-stream restoration options and apply the hierarchical strategy of Roni et al. (2002) to prioritize among them. That strategy emphasizes habitat reconnection as generally the most effective and certain of in-stream strategies, where prior disconnection is among the problems. The strategy then guides a user through consideration of riparian restoration and road improvements, with in-stream structural placements to follow or occur simultaneously with any of the other actions, as appropriate.

Strategies for Wetlands Management

Introduction

Over the past half century it has been recognized that wetlands, once thought to be nuisances, perform functions beneficial to the broader environment and its inhabitants, including humans. Functions are defined as the physical, biological, chemical, and geologic interactions among different components of the environment that occur within a wetland. There are many valuable functions that wetlands perform but they can be grouped into three broad categories: functions that improve water quality, functions that change the water regime in a watershed, and functions that provide habitat for plants and animals (Box 5)(Sheldon et al. 2005).

Box 5. Examples of wetland ecological functions (from Sheldon et al. 2005)

- Capture pollutants that would otherwise travel farther;
- Store flood waters;
- Recharge groundwater in some cases and provide a passageway for groundwater to supplement surface flows in others;
- Produce food through plant photosynthesis that is often exported to nourish downstream waters; and
- Provide breeding grounds, feeding sites, and sometimes permanent homes for invertebrates, amphibians, birds, and mammals.

A great deal of work has been devoted to methods to establish the type and extent of the functions performed by wetlands. This work culminated in the development of the Hydrogeomorphic Method (HGM, Brinson 1993), which classifies a wetland based on its landscape setting, water source, and internal water dynamics. HGM was adapted for regional application by the Washington State Wetlands Function Assessment Project[5]

As a consequence of their functions, threats to wetlands become threats not only to their internal ecosystems but also to waters and even terrestrial environments associated with them throughout the Puget Sound ecosystem. Therefore, protection and restoration of wetlands very broadly counters threats associated with their many functions (see Box 5). In turn, strategies to protect and restore wetlands support an array of broader strategies, essentially all of those discussed earlier in this chapter because of the intimate association of wetlands with streams. Protecting or recovering specific functions, at particular levels, is the natural basis for setting wetland restoration objectives.

While it is obvious that creating something new is generally a harder task than improving something already at least partially functioning, success in protecting, restoring, and creating wetlands depends on the principles put forth by Sheldon et al. (2005): the system must have the structural elements needed to support the intended functions. These elements are the general climate, the geomorphology (topography, landforms, soils, and geology), the source of water, and the movement of water. These factors affect wetland functions directly or through a series of secondary factors, including nutrients, salts, toxic contaminants, temperature, and the connections created between different patches of habitat (Sheldon et al. 2005).

Here we employ the general term "wetlands management" to refer to protection, restoration, and creation of wetlands, when the subject pertains to all of those facets. This selection of terminology is because of the overlapping nature of many considerations for success in any of these areas and a desire to avoid repetition of the three in many places.

Wetlands science and management are very well developed in Washington State. The Washington Department of Ecology (WDOE) has had an active wetlands program for many years. WDOE, with other partners, has supported extensive research to advance the science on specific questions and inform its management and regulatory efforts. The program has assimilated the results of this regional research, as well as findings from the broader literature, to report on virtually every aspect of wetlands science and management pertinent to this valuable Puget Sound resource. WDOE went on to develop guidance for its own and local government staff and private parties regarding the protection and restoration of wetlands and mitigation of their losses. The resulting documents total many hundreds of pages rich in information. More important than their length is their rooting in "best available science." Recapping this record is beyond the scope of this review and, moreover, is unnecessary given its effective reporting in the source documents. Accordingly, the review is confined to drawing out strategies related to the PSP Action Agenda and Results Chain memorandum and pointing out supporting material in these documents.

A major study performed under WDOE and other sponsorship was the Puget Sound Wetlands and Stormwater Management Research Program. This work spanned 11 years and produced numerous papers and a book recounting the entire process and its results, conclusions, and recommendations (Azous and Horner 2001). It was designed similar to the stream research covered extensively earlier, in that physical, chemical, and biological variables were measured in wetlands subject to a broad range of watershed urban development.

Wetlands Management Guidance

Two principal documents published by the Washington Department of Ecology provide background and guide protection and restoration of Puget Sound's wetlands (Sheldon et al. 2005, Granger et al. 2005). First, Sheldon et al. (2005) describe the state's wetlands and how they function. The authors elucidate how human activities disturb wetlands and the resulting negative impacts on their functioning. They then turn to the science behind wetlands management, including mitigation, and the effectiveness of the available tools. The management section emphasizes wetland buffers, relatively undisturbed surround lands offering important protective benefits. They also examine cumulative impacts and recommended responses to counter them. Second, Granger et al. (2005) lay out a framework for managing wetlands using best available science. They emphasize that the first step is analyzing wetlands in a landscape context, which they frame as a series of questions with guidance to obtaining the needed answers (Granger et al. 2005). Individual chapters cover regulatory and non-regulatory solutions to reduce and risks from human activities. They conclude by discussing implementation, monitoring, and adaptive management (Granger et al. 2005).

For mitigating wetlands losses, two additional, more specialized documents are available from WDOE: "*Wetland Mitigation in Washington State, Part 1: Agency Policies and Guidance" (WDOE 2006a) and "Wetland Mitigation in Washington State, Part 2: Developing Mitigation Plans" (WDOE 2006b)*. Part 1 explains the regulatory requirements, and Part 2 provides a more qualitative and descriptive basis for developing mitigation plans. Designed for use by qualified and experienced technical experts, the two-part series is topically comprehensive to bring in the full range of considerations that must be considered to produce effective mitigation for losses of wetland area and functions including mechanisms like mitigation banking and in lieu payments to compensate for losses. More general information is contained within *Wetlands* by Mitsch and Gosselink (2007) and its companion volume, *Wetland Ecosystems* by Mitsch et al. (2009).

Effectiveness and Relative Certainty of Wetlands Management Efforts

Research Basis for Effectiveness and Certainty Assessment

The Puget Sound Wetlands and Stormwater Management Research Program has performed research with the goal of deriving strategies that protect wetland resources in urban and urbanizing areas, while also benefiting the management of urban stormwater runoff that can affect those resources (Chin 1996, Horner et al. 1997, Reinelt et al. 1998, Azous and Horner 2001). The research consisted of long-term comparisons of wetland ecosystem characteristics before and after their watersheds wereurbanized, and between a set of wetlands that became affected by urbanization (treatment sites) and a set whose watersheds did not change (control sites). This work was supplemented by shorter term and more intensive studies of pollutant transport and fate in wetlands and several laboratory experiments. These research efforts were aimed at defining the types of impacts that urbanization can cause and the degree to which they develop under different conditions, in order to identify means of avoiding or minimizing impacts that impair wetland structure and functioning. The program's scope embraced both situations where urban drainage incidentally affects wetlands in its path, as well as those in which direct stormwater management actions change wetlands' hydrology, water quality or both.

A major finding of the Puget Sound Wetlands and Stormwater Management Research Program was a decline in the biotic diversity of wetlands associated with increase in water level fluctuations (WLF) and increasing total impervious area (TIA) within the contributing basins (Chin 1996, Horner et al. 1997, Reinelt et al. 1998, Azous and Horner 2001) (see Appendix 4C for supporting material).

Horner et al. (1997) found that an increase of mean annual WLF above 20 cm and a near certainty with TIA > 21 percent resulted in significant decreases in wetland biodiversity (Horner et al. 1997). WLF above 22cm was also negatively correlated with vegetation species richness in emergent vegetation and scrub-scrub wetland habitats. Species richness for both wetland plants and amphibians in wetlands exhibited the same trends as invertebrates and fish in streams with respect to watershed urbanization such that higher levels of wetland health were observed only in watersheds with less urbanization but allow-urbanization sites did not necessarily have healthy wetlands; on the other hand, a lower levels of biological integrity were consistently observed at high levels of urbanization (Horner et al. 1997).

Wetland “hydroperiod” comprises not only the extent but also the frequency and duration of water level fluctuations. In wetlands studied before and after urbanization increased, the frequency and duration of excursions (deviations from) a certain level above or below the pre-existing mean water level were also associated with biodiversity decline (Azous and Horner 2001). Marked plant species richness decrease was seen whenmore than six excursions per year were > 15 cm above or below the pre-existing level, and with any excursion of that magnitude lasting more than 72 hours. Those conditions occurred in the majority of cases when mean annual WLF rose above 24 cm and 20 cm, respectively. Fewer plant species were also recorded if the summer dry period increased or decreased by more than two weeks from the pre-existing length. For amphibians, decrease in species numbers occurred with excursions of > 8 cm for more than 24 hours in any 30 day period in the breeding season (February 1 to May 31) (Azous and Horner 2001).

Other research efforts have found wetland hydrology and hydrodynamics to be functions not only of characteristics of the contributing watershed but also of three aspects of the wetland geomorphology: (1) hydrodynamic type (open water or flow through), (2) outlet constriction (high, moderate, or low), and (3) wetland-to-watershed area ratio (Reinelt et al. 2001, Reinelt and Taylor 2001). Wetlands were classified as the open-water type if substantial pools without emergent vegetation were present, channelization was largely absent, and water velocities were predominantly low (< 5 cm/s). High outlet constriction was characterized by having an undersized culvert or confining beaver dam or by being a completely closed depression. Low constriction was marked by free discharge as sheet flow, over a broad bulkhead, or via an oversized culvert. Wetlands small in area compared to their watersheds (comprising < 5 percent of total area) tend to be dominated by surface inflows, whereas groundwater influence assumes greater importance with relative enlargement of the wetland. Outlet constriction was found to the most important geomorphic variable in controlling WLF (Reinlet and Taylor 2001).

Another major result of the Puget Sound Wetlands and Stormwater Management Research Program was cataloging the maximum, median, and minimum water levels at which nearly 100 vegetation species were found (Cooke and Azous 2001; for details see Appendix 4C, Table C1). While many species occurred over a wide hydrologic range, others were only observed within a narrow range.

Wetland management guidelines were formulated from these and other results of the Puget Sound Wetlands and Stormwater Management Research Program (Horner et al. 2001) and were adopted into the Washington Department of Ecology's Stormwater Management Manual for Western Washington (WDOE 2005). Wetland protection efforts, as well as attempts to restore wetlands or create new ones, can be designed with the use of this information; and their prospective effectiveness and its relative certainty can be judged accordingly. Another important feature of the guidelines was advice on protecting wetlands from adverse impacts resulting from altering the quantity or quality of entering water (WDOE 2005).

Reported Wetland Mitigation Effectiveness

The Washington State Wetland Mitigation Evaluation Study (Johnson et al. 2000, Johnson et al. 2002) was developed in two phases to evaluate the success of projects intended to compensate (mitigate) for wetlands lost to development activities in the state of Washington. In the first phase of the study Johnson et al. (2000) examined the compliance of 45 randomly selected projects with their permit requirements. Permit compliance for each of the 45 compensatory wetland mitigation projects was evaluated relative to three questions: 1) Was the compensatory mitigation project implemented, 2) Was it implemented according to plan. and 3) Was it meeting its performance standards. They found that overall, 29 percent of projects were in full compliance with all three criteria (Johnson et al. 2000). Forty-two projects (93 percent) were implemented, and of those, 55 percent were implemented according to plan. However, only 35 percent were meeting all performance standards (Johnson et al. 2000).

In the second phase of the study, Johnson et al. (2002) examined the ecological functions of a subset of 24 projects. The ecological success of mitigation projects was evaluated based on two factors, each with its own criteria: First, achievement of ecologically relevant measures (establishing the required wetland area, attaining ecologically significant performance standards, and fulfilling goals and/or objectives); and second, adequate compensation for the loss of wetlands (contribution of the mitigation activity to the potential performance of functions, comparison of the type and scale of functions provided by the mitigation projectwith the type and scale of lost wetland functions). They found that based on these criteria, only 13 percent of the projects were judged to be fully successful, 33 percent were moderately successful, 33 percent were minimally successful, and 21 percent were unsuccessful. Created wetlands were found to be more successful thanprevious studies had shown, with 60 percent at least moderatelysuccessful and only one project unsuccessful. In contrast, no enhancement projects were fully successful, while eight out of nine (89 percent) enhanced wetlands were minimally successful or unsuccessful. For wetland restoration and creation projects together only 65 percent ofthe total acreage of wetlands lost was replaced (Johnson et al. 2002).

The NRC committee on compensating for national wetlands losses reported that from 1993 to 2000 approximately 24,000 acres of wetlands were permitted for filling, and around 42,000 acres of compensation was specified (NRC 2001). However, because of lack of recorded data, the committee was unable to establish the original wetland functions lost or even how much of the specified area was actually restored or created. Nevertheless, their investigations produced a number recommendations that if adopted would likely improve mitigation success. Key scientific recommendations relative to wetland functioning, in summary form are listed in Box 6.

Box 6. Recommendations for improving mitigation success from NRC (2001)

- Wetland conservation and mitigation should be pursued on a watershed scale.
- Biological dynamics should be evaluated in terms of population present in reference wetlands for the region and the ecological requirements of those species.
- Hydrologic functioning should be incorporated into mitigation design and should also be based on comparison to reference sites.
- Provide appropriate, heterogeneous topography.
- When establishing vegetation, pay particular attention to planting elevation, depth, seasonal timing, and soil.
- Mitigation goals must be clear, and those goals should be carefully specified in terms of measurable performance standards.
- Third-party compensation approaches (e.g., mitigation banks, in lieu fee programs) offer some advantages over permittee-responsible mitigation and should be further evaluated according to a taxonomy developed by the committee.

Climate Change Implications for Wetlands

The anticipated results of climate change covered earlier with reference to streams also have implications for wetlands. The expected shift to more precipitation in the cooler, wetter season and less in the warmer, drier season was projected to produce proportionately greater runoff increases, because of the higher efficiency of cool-season precipitation in producing runoff, and higher cool-season and lower warm-season stream flows (Elsner et al. 2009). These effects could increase the magnitude, frequency, and duration of wetland water level fluctuations. To the extent these increases alter conditions outside their preferred and tolerated ranges, wetland plant and amphibian communities could be affected, generally in the direction of reduced species richness. Such occurrences would make more difficult the protection of existing biodiversity and wetland functions and restoration efforts to recover higher levels.

The projected higher summer temperatures and lower runoff would together tend to increase wetland drying, perhaps introducing a dry period in wetlands where one previously did not happen and increasing its length in those already experiencing one. As the research revealed, plant species presence is likely to be lower with any lengthening beyond two weeks (Azous and Horner 2001).

Synthesis of Strategies for Wetlands Management

The two WDOE volumes covering wetlands management overall (WDOE 2006a, WDOE 2006b) and two additional mitigation documents (Johnson et al. 2000 and Johnson et al 2002), introduced earlier, provide guiding frameworks to implement wetlands protection, restoration, and creation strategies. These works can be supplemented by specific results from the literature and the expertise to put them into appropriate use to achieve the set objectives. The Mitsch and Gosselink (2007) text, the HGM-based functional assessment procedure built into WDOE's program, the recommendations from NRC (2001), and the Puget Sound Wetlands and Stormwater Management Research Program results reported here are foundations for designing and implementing strategies to manage Puget Sound's wetlands to advance the PSP's program.

Together, these studies demonstrate that the physical tolerances of target biological communities must be met in orderto retain or recover these communities, which can only occur if the major factors controlling those influences (climate, geomorpology, and the source and movement of water) are consistent with supplying them. Urbanization and the projected climate change effects described here will clearly have a bearing and must be considered in strategies for wetland protection, restoration, and creation.

Importantly, the body of research we discuss here points to the primacy of hydrologic and hydroperiod preferences and tolerances in governing community maintenance and development and provides specific quantitative specifications supportive of plants and amphibians (e.g., Chin 1996, Horner et al. 1997, Reinelt et al. 1998, Azous and Horner 2001). Establishing hydrologic and hydroperiod variables usually requires the use of computerized models capable of continuous pattern simulations based on meteorological input data, especially for restoration and wetland creation projects. The Western Washington Hydrologic Model (WDOE 2005) is one such model available for this task. Exerting control over those variables will require the application of stormwater management strategies described later in this chapter.

As pointed out above, wetland geomorphology includes the topography, landforms, soils, and geology (Reinelt et al. 2001, Reinelt and Taylor 2001). Scientists and engineers working to protect, restore, or create wetlands have a relatively high degree of control over the first two of those variables. They also have some influence over soils, which can be altered in wetland restoration and creation projects through mineral and organic amendments.

The general topography of a wetland is best represented by its side slopes, which tend to be quite gradual (e.g., ~12 horizontal:1 vertical) in natural wetlands. Wetlands have often been created on more of a "farm pond" model with much steeper side slopes (e.g., ~4 horizontal:1 vertical). If these two configurations received identical inflow volumes, the resulting depth would be much greater (on the order of twice, depending on dimensions) in the latter compared to the first case. It is highly likely in this scenario that the preferences and tolerances of some biota present in the first wetland would not be reproduced in the second, meaning those organisms probably would not thrive there or would be absent entirely. This single factor is probably the leading explanation for some of the numerous documented wetland mitigation failures. A strategy of designing structural features of created and restored wetlands according to those seen in one or more reference natural wetlands can avoid these errors.

Hydrological features such as outlet constriction, pool-and-channel pattern, and wetland-to-watershed area ratio have all been demonstrated toaffect hydrology and hydroperiod (Reinelt et al. 2001, Reinelt and Taylor 2001). If the outlet is unfavorable to internal conditions, it is relatively easy to change it, to protect existing wetland functions, or design one conducive to the functional objectives being pursued in a restoration or creation project. The latter activities also give latitude to form or reform the internal structure. There is less possible control over the area ratio, but designers must be cognizant of its strong influence on hydrology, quantify that influence properly, and design to manage it in relation to objectives.

Key Strategy: Protect, restore, and create wetlands according to the known preferences and tolerances of target biological communities, particularly geomorphic, hydrological, and hydroperiod requirements.

Strategies for Management of Lakes

As the location of one of the most well-known lake eutrophication and recovery episodes in the world, the Puget Sound area has long been a center of research on the negative effects of waste discharges to lakes, the resulting deterioration, and restoration strategies. Lake Washington had received discharges from municipal wastewater treatment plants around its shore for approximately 10 years when its deteriorating condition became increasingly evident. Two references that generally describe the recovery process and the shift in nutrient dynamics, algal populations and water clarity are Cooke et al. (2005) and Welch and Jacoby (2004). The entire decline and recovery was meticulously studied and documented by Thomas Edmondson and his associates at the University of Washington and provided some of the most complete and conclusive early evidence of the role of phosphorus in lake trophic (nutritional) dynamics.

Lake Washington's recovery was rapid for several reasons: its relatively great depth, rapid flushing rate, short history of enrichment, and the low phosphorus content of its major tributary, the Cedar River. These circumstances are not typical, and neither is the pattern of recovery. Lake Sammamish also went through a eutrophication episode, related to a municipal treatment plant and dairy waste, and recovered more slowly because of differing conditions but eventually to about the same degree. This experience, and others in many lakes, produced sufficient understanding that the results of changing the enrichment of a lake, in either the positive or negative direction, are fairly predictable. A number of restoration techniques exist. While a lake's particular situation determines the best selection and its prospects for success, the state of lake science is such that a strong basis exists to assess the situation, select a technique, and forecast the results with a greater degree of certainty than often is the case with natural ecosystems.

Excellent resources exist to guide lake analysis and restoration assessment. Because of the extensive work here, they have a strong regional flavor. Among many volumes that could be consulted for analysis, Welch and Jacoby (2004), and specifically its Part II on Effects of Pollutants in Standing Water, is recommended for comprehensiveness, practicality, and blend of regional and worldwide content. Chapter 7 (Eutrophication) provides straightforward mathematical expressions and criteria to classify a lake's trophic status and begin to make decisions about its management and possible restoration. Cooke et al. (2005) provide the detailed science on lake restoration, covering the full range of methods with a similar approach, numerous case studies, and coverage of issues of effectiveness and relative certainty. They divide their presentation of restoration strategies into algal biomass control, control of macrophytes (emergent, submergent, or floating aquatic plants), and treatments for multiple benefits (see Appdendix 4D, Box D1 for details of algal biomass control) and form the basis for the following key strategy: *Key Strategy: Protect and restore lakes applying the established specific strategies of algal biomass and macrophyte control.*

Urban Stormwater Management Strategies

Introduction

Stormwater runoff in general, and the component of that runoff from urban lands in particular, has emerged as an issue of widespread concern in the Puget Sound region since urban stormwater runoff is a major source of toxic chemicals in the marine waters of Puget Sound (HartCrowser 2007, Envirovision 2008, WDOE 2008, PSP 2010).

Both national (e.g., the Nationwide Urban Runoff Program [NURP], USEPA 1983) and regional research (e.g., Mar et al. 1982) have shown that storm runoff from developed lands threatens not only flooding but also the water quality of streams, lakes, and marine waters receiving the discharge. With their roots in flood control engineering, it was natural than early stormwater managers would turn to structural solutions to the newly recognized problems. Holding runoff in a detention pond for a time and releasing it more slowly than flow off urban surfaces became the favored response to reduce peak flow rates, and hence the erosive shear stress built up in streams. For water quality improvement USEPA's (1983) NURP highlighted "wet ponds," basins that maintain a permanent (or semi-permanent) pool and hold runoff for an extended period, giving pollutant removal mechanisms an opportunity to function. These measures prevailed over the first 20 years of modern stormwater management, until ideas about "low-impact development" began to coalesce, at first in Maryland (Prince George's County 1999). As explained in much more detail below, low-impact development (LID) is a system of practices aimed at avoiding runoff above pre-development quantities and its contaminants from being generated in the first place or preventing it from discharging off-site by exploiting the capabilities of vegetation and soil to mimic pre-development site hydrology. This approach contrasts with traditional methods, which are oriented more toward structural end-of-pipe control.

A National Academy of Sciences committee recently reviewed the entire history and status of the nation's urban stormwater management program. The committee's report (NRC 2009) identified numerous problems with the program and made many corrective recommendations, some of which are listed in Box 7. These recommendations lay the groundwork for the specification of detailed strategies to counter the multiple negative effects of urban stormwater runoff on Puget Sound and its tributary waters.

Box 7. Recommendations from NRC (2009) for urban stormwater management:

- Individual controls on stormwater discharges are inadequate as the sole solution to stormwater in urban watersheds. Storm-water control measure (SCM)[6] implementation need to be designed as a system, integrating structural and nonstructural SCMs and incorporating watershed goals, site characteristics, development land use, construction erosion and sedimentation control, aesthetics, monitoring, and maintenance.

- Nonstructural SCMs such as product substitution, better site design, downspout disconnection, conservation of natural areas, and watershed and land-use planning can dramatically reduce the volume of runoff and pollutant load from a new development. Such SCMs should be considered first before structural practices.
- SCMs that harvest, infiltrate, and evapotranspire stormwater are critical to reducing the volume and pollutant loading of small storms.

The most effective solutions are expected to lie in isolating, to the extent possible, receiving water bodies from exposure to impacts such as stream channel modification, degradation of salmon spawning and rearing habitat, disruption of the stream food web, exposure to toxic pollutants, and increased pollutant loading to downstream waters including the Puget Sound. In particular, low-impact design methods, termed Aquatic Resources Conservation Design (ARCD) in this report, should be employed to the fullest extent feasible and backed by conventional SCMs when necessary.

Here we first explore the most highly recommended LID-based strategies followed by the conventional end-of-pipe practices. Available effectiveness and relative certainty data are presented for both strategy categories. This account supports development of PSP Results Chain strategies for flow protection and stormwater control enumerated in Neuman et al. (2009).

Aquatic Resources Conservation Design (ARCD) Strategies[7]

Aquatic Resources Conservation Design (ARCD) Strategies (NRC (2009) are more encompassing than Low-Impact Development (LID) because ARCD signifies that the principles and many of the methods apply to both building on previously undeveloped sites as well as redeveloping and retrofitting existing development. Additionally, incorporating aquatic resources conservation helps reinforce the main reason for improving stormwater regulation and management. ARCD encompasses the all practices that can be used to reduce negative imp1acts of urban runoff ; i.e., "… the full suite of practices that decrease surface runoff peak flow rates, volumes, and elevated flow durations, as well as those that avoid or at least minimize the introduction of pollutants to any surface runoff produced" (NRC 2009, p. 406). Reducing the concentration of pollutants and volume of runoff reduces the cumulative amount of pollutants released into receiving waters.

According to the NRC (2009) report "…ARCD practices begin with conserving existing vegetation and soils, as well as natural drainage features (e.g., depressions, dispersed sheet flows, swales). Clustering development to affect less land is a fundamental practice advancing this goal. Conserving natural features would further entail performing construction in such a way that vegetation and soils are not needlessly disturbed and soils are not compacted by heavy equipment. Using less of polluting materials, isolating contaminating materials and activities from contacting rainfall or runoff, and reducing the introduction of irrigation and other non-stormwater flows into storm drain systems are essential. Many ARCD practices fall into the category of minimizing impervious areas through decreasing building footprints and restricting the widths of streets and other pavements to the minimums necessary. Water can also be harvested from impervious surfaces, especially roofs, and put to use for irrigation and gray water system supply. Harvesting is feasible at the small scale using rain barrels and at larger scales using larger collection cisterns and piping systems. Relatively low traffic areas can be constructed with permeable surfaces such as porous asphalt, open-graded Portland cement concrete, coarse granular materials, concrete or plastic unit pavers, or plastic grid systems. Another important category of ARCD practices involves draining runoff from roofs and pavements onto pervious areas, where all or much can infiltrate or evaporate in many situations" (NRC 2009, p.407).

Following the initial application of ARCD, any excess site runoff could be a candidate an array of techniques to reduce the quantity through infiltration and evapotranspiration (ET) and improve the quality of any remaining runoff. "Natural soils sometimes do not provide sufficient short-term storage and hydraulic conductivity for effective surface runoff reduction because of their composition but, unless they are very coarse sands or fine clays, can usually be amended with organic compost to serve well" (NRC 2009, p. 407).

The NRC (2009) report recommends that "ARCD practices be designed to be applied as close to sources as possible to stem runoff and pollutant production near the point of potential generation. However, these practices must also work well together and, in many cases, must be supplemented with strategies operating farther downstream. For example, the City of Seattle, in its "natural drainage system" retrofit initiative, built serial bioretention cells flanking relatively flat streets that subsequently drain to "cascades" of vegetated stepped pools created by weirs were installed along more sloping.streets. The upstream components are highly effective in attenuating most or even all runoff. Flowing at higher velocities, the cascades do not perform at such a high level, although under favorable conditions they can still infiltrate or evapotranspire the majority of the incoming runoff (Horner et al. 2001, 2002, 2004, Chapman 2006, Chapman and Horner 2010). [The cascades] extract pollutants from remnant runoff through mechanisms mediated by vegetation and soils. The success of Seattle's natural drainage systems demonstrates that well designed ARCD practices can mimic natural landscapes hydrologically, and thereby avoid raising discharge quantities above pre-development levels." (NRC 2009, p407)

In cases where ARCD approaches are not feasible, conventional SCMs such as retention/detention basins, biofiltration, and sand filters can be used to augment ARCD strategies. As pointed out in the NRC (2009) report, a "…watershed-based program emphasizing ARCD practices would convey benefits beyond improved stormwater management. ARCD techniques overall would advance water conservation, and infiltrative practices would increase recharge of the groundwater resource. ARCD practices can be made attractive and thereby improve neighborhood aesthetics and property values. Retention of more natural vegetation would both save wildlife habitat and provide recreational opportunities. Municipalities could use the program in their general urban improvement initiatives, giving incentives to property owners to contribute to goals in that area while also protecting water resources." (NRC 2009, p. 407-408)

ARCD practices are numerous and expanding as existing configurations are applied in new ways. Table 5 presents a catalogue adapted from USEPA (2007b) and NRC (2009). This catalogue contains practices that are not equally applicable in all settings; e.g., residential, industrial, and commercial land uses; or new development, redevelopment, and retrofit stages. Nevertheless, each category offers practices applicable in each of these circumstances.

Table 5. A Catalogue of Aquatic Resources Conservation Design Practices, adapted from USEPA (2007b) and NRC (2009)

Category	Definition	Examples
Source control	Minimizing pollutants or isolating them from contact with rainfall or runoff	• Substituting less for more polluting products • Segregating, covering, containing, and/or enclosing pollutant-generating materials, wastes, and activities • Avoiding or minimizing fertilizer and pesticide applications • Removing animal wastes deposited outdoors • Conserving water to reduce non-stormwater discharges
Conservation site design	Minimizing the generation of runoff by preserving open space and reducing the amount of land disturbance and impervious surface	• Cluster development • Preserving wetlands, riparian areas, forested tracts, and porous soils • Reduced pavement widths (streets, sidewalks, driveways, parking lot aisles) • Reduced building footprints
Conservation construction	Retaining vegetation and avoiding removing topsoil or compacting soil	• Minimizing site clearing • Minimizing site grading • Prohibiting heavy vehicles from driving anywhere unnecessary
Runoff harvesting	Capturing rainwater, generally from roofs, for a beneficial use	• Storage and distribution system for gray water and/or irrigation supply for a public building • Cistern for residential garden watering
Category	Definition	• Examples
Practices for temporary runoff storage followed by infiltration and/or evapotranspiration[a]	Use of soil pore space and vegetative tissue to increase the opportunity for runoff to percolate to groundwater or vaporize to the atmosphere	• Bioretention cell (rain garden) • Vegetated swale (channel flow) • Vegetated filter strip (sheet flow) • Planter box • Tree pit • Infiltration basin • Infiltration trench • Roof downspout surface or subsurface dispersal • Permeable pavement • Vegetated (green) roof
ARCD landscaping[b]	Soil amendment and/or plant selection to increase storage, infiltration, and evapotranspiration	• Organic compost soil amendment • Native, drought-tolerant plantings • Reforestation • Turf conversion to meadow, shrubs, and/or trees

[a]Some of these practices are also conventional stormwater BMPs but are ARCD practices when ARCD landscaping methods are employed as necessary to maximize storage, infiltration, and evapotranspiration. The first five examples can be constructed with an impermeable liner and an underdrain connection to a storm sewer, if there is a good reason to do so (see further discussion later). Vegetated roofs store and evapotranspire water but offer no infiltration opportunity, unless their discharge is directed to a secondary, ground-based facility.

[b]Selection of landscaping methods depends on the ARCD practice to which it applies and the stormwater management objectives, but amending soils unless they are highly infiltrative and planting several vegetation canopy layers (e.g., herbaceous growth, shrubs, and trees) are generally conducive to increasing storage, infiltration, and evapotranspiration.

The best strategy for choosing among and implementing these practices is a decentralized, integrated one; i.e., selecting practices that fit together as a system, working from at or near sources through the landscape until management objectives are met. This strategy makes maximum possible use of practices in source control, conservation site design and conservation construction, which then can help prevent stormwater quantity and quality problem. Source control and preservation of existing vegetation and soils obviously avoid post-development runoff quantity and prevent pollutant runoff. Among all strategies, these best maintain pre-development hydrology (infiltration and ET patterns) and yield of materials flowing from the site. This preventive strategy is supplemented by creating as little impervious cover as possible. The remaining practices then contend with the excess runoff and pollutants over pre-development levels generated by the development.

For the practices that infiltrate water, a site's infiltration capability can be determined through infiltration rate testing and excavation to determine the pattern of soil layers and if groundwater approaches the surface too closely for effective operation. Because of the often substantial variability of conditions around a site, these determinations need to be made at multiple points. Guidance cited below provides procedures for these tasks. If the natural infiltration rate is insufficient, generally regarded as < 0.5 inch/hour (< 1.25 cm/h, Geosyntec 2008), in many situations the soil can be amended, usually with organic compost, to apply an infiltrative practice.

Less predictable than infiltration at this point in time is evapotranspiration. Evidence gathering from available performance research, presented later, is that ET can be substantial but is difficult to quantify at a given site without more research. Therefore, designs of these facilities cannot be optimized now for maximum performance. Meanwhile, designing on the basis of infiltration rate, set considering soil amendment if any, is conservative and is likely to yield better than predicted performance as a result of ET.

Strategies for Implementing ARCD Practices

No single manual on LID provides up-to-date comprehensive selection, design, installation, and maintenance advice covering the full range of practices listed in Table 5. Both the USEPA[8] and the Center of Low Impact Development[9] websites list several manuals, but they are somewhat dated and focused on specialized applications.

Regionally, the Puget Sound Action Team (PSAT) produced a guidance manual (Hinman 2005) that emphasized site design and construction practices most appropriate for residential land use. This manual guides site assessment and planning, covering conservation construction, soil amendment, and aspects of conservation site design; and provides detailed specifications for bioretention cells, permeable pavement, vegetated roofs, and roof water harvesting. Hinman (2007) supplemented the manual with a handbook aimed at homeowners who wish to create small-scale rain gardens.

Further information on ARCD can be gained from the Post-Construction BMP Technical Guidance Manual of the City of Santa Barbara, California (Geosyntec Consultants 2008), which emphasizes ARCD techniques over conventional ones. This manual encompasses many major urban land use categories although it does not cover source control. Volume IV of WDOE's (2005) Stormwater Management Manual for Western Washington fills that gap well for commercial and industrial areas. Box 8 provides other resources on various aspects of ARCD.

Box 8. Aspects of ARCD of interest to Puget Sound practitioners.

- Green Roofs for Stormwater Runoff Control (Berghage et al. 2009)—report on comprehensive studies of green roof performance at Pennsylvania State University, with recommendations pertaining to future designs;
- Innovative Approaches for Urban Watershed Wet-Weather Flow Management and Control: State-of-the-Technology: Interim Report (Struck, Rowney, and Pechacek 2009)—presents a global information search to identify state-of-the-technology approaches, including case examples and conclusions and recommendations to guide future research, development, and demonstration initiatives;
- Managing Wet Weather with Green Infrastructure Action Strategy (USEPA 2008)— an outline of a national strategy to develop and implement ARCD; and
- Reducing Stormwater Costs through Low Impact Development (LID) Strategies and Practices (USEPA 2007b)—presents five case studies from the Pacific Northwest and 12 more nationally with design, performance, and cost information.

See Appendix 4E for supporting information on ARCD strategies and stages of urbanization

Effectiveness and Relative Certainty of ARCD Strategies

Most of the ARCD practices in Table 5 are preventive; i.e., they avoid the generation of surface runoff above pre-development levels and additions of pollutants over pre-development amounts to whatever surface runoff still occurs. If these practices are applied effectively, they could be 100 percent effective for any land area covered. For example, if a development is clustered and an existing forest is untouched by it, the forest's runoff production and characteristics should not change. Likewise, shielding a previously outdoor industrial activity from contact with rainfall and runoff would eliminate the contamination from that operation. The question of effectiveness and relative certainty, therefore, apply mainly to the last two groups of practices in the table, those for temporary runoff storage followed by infiltration and/or evapotranspiration, and the associated ARCD landscaping practices.

To express hydrologic performance, the most common measures seen in the literature are cumulative surface runoff volume reduction between inflow and outflow over a period that includes multiple storms, and peak flow rate reduction statistics. In their recent paper on the performance of one of Seattle's natural drainage systems, Chapman and Horner (2010) reported water quality performance in terms of long-term pollutant mass loading reductions, reliable effluent pollutant concentrations, and irreducible minimum effluent concentrations. Mass loading represents a cumulative burden on the receiving water, while the maximum expected discharge concentration expresses the highest acute stress exerted on aquatic life. The irreducible minimum concentration indicates the best performance that can be expected. It is now well established that quantifying stormwater BMP performance in relation to concentration reduction (influent to effluent) statistics can be misleading for several reasons (Strecker et al. 2001), a key one being that devices are often observed to put out somewhat consistent effluent pollutant levels in the face of relatively variable influent levels (Barrett 2005). In that situation a device receiving a relatively "cleaner" flow would not register a performance efficiency as high as one getting a "dirtier" influent, even if their effluent concentrations are identical.

Seattle Natural Drainage System Effectiveness and Relative Certainty

There has been more study of some form of bioretention than any other ARCD practice. This method is applied in a variety of configurations, in some cases with elements of swales and filter strips in addition to or even instead of a depressional form. The City of Seattle's natural drainage system program[10] exemplifies this approach. As described earlier, Seattle uses two basic models: serial bioretention cells for relatively flat streets, and "cascades" of vegetated stepped pools created by weirs along more sloping streets. The former are quiescent, while the cascades often have flow at visible velocity, and hence are "swale-like."

The best known flat-street cellular installation is the 2nd Avenue Northwest Street Edge Alternatives (SEA Streets) project. This project's performance has been widely reported regionally, nationally, and internationally, although not in any peer-reviewed journal form because of the straightforwardness of the results. The following account comes from a report to the city by the University of Washington (Horner and Chapman 2007). The street was redesigned to reduce impervious cover, and also traffic speeds, while converting previous asphalt and gravel right of way to vegetated swales and detention areas. Built largely in compost-amended soils, this natural drainage system was designed to reduce peak runoff rates and volumes conveyed to Pipers Creek. While providing these environmental benefits, the system landscaping was also intended to offer a neighborhood aesthetic benefit.

Prior to construction of the SEA Streets project baseline flow monitoring occurred during the period March 19-June 18, 2000 and embraced 35 events totaling 6.32 inches (161 mm) of precipitation. The catchment discharged in all events, delivering a total of 8601 ft3 (244 m^3) of runoff to the downstream drainage system leading to Pipers Creek. As a crude measure of yield, the street generated 1361 ft^3 of runoff per inch of rain (1.52 m^3 per mm).

Monitoring of the completed SEA Streets project began on January 20, 2001. Over the next approximately two years (through March 31, 2003) the system experienced 162 events producing 76.9 inches (1954 mm) of precipitation. The new street discharged runoff during only 11 storms

(6.8 percent), yielding 1948 ft3 (55 m^3) of runoff, or 25.3 ft^3 of runoff per inch of rain (0.028 m^3 per mm). This yield is just 1.9 percent of the amount before the project's construction.

Flow monitoring continued through June 30, 2007. The last recorded discharge was on December 14, 2002. On and about October 20, 2003 the Seattle-Tacoma International Airport rain gauge registered its highest ever 24-hour rainfall total. The Viewlands rain station in the 2nd Avenue NW neighborhood recorded 4.22 inches (107 mm) of rain from late on October 19, 2003 to the morning of October 21 (a period of 32.5 hours). The next month a quantity of 3.86 inches (98 mm) fell at Viewlands over a 51.25-hour period from November 17 to 19, 2003. Then, in November 2006 Seattle experienced its largest ever monthly rainfall, 15.63 inches (397 mm) at the airport. Therefore, the SEA Streets drainage system ceased discharging runoff even with exposure to large short- and long-term precipitation quantities.

The 2nd Avenue NW SEA Streets site thus demonstrated a clear tendency to store and prevent surface runoff from even more rainfall than during its early years. The reason for this development can only be speculation. However, it is likely that the vegetation, as it matures, more effectively intercepts rainfall, after which it can evaporate; assimilates more water into its tissues, for storage and possible transpiration; and assists percolation through the soil by piping water along the root structures. In this condition the bioretention unit mimics a natural Pacific Northwest landscape, in which surface runoff is unusual.

The most complete performance study of a cascade was performed on the NW 110th Street system (Chapman and Horner 2010). It has 5-8 cm of 4-cm-diameter washed gravel over a minimum 20-cm-deep layer of soil mix containing 30 percent organic compost and 70 percent gravelly sand (by volume). These amended soils are underlain by a layer of 6-mm-diameter, bank-run gravel. Over three full wet seasons and two dry seasons, 235 storms delivered 2.23 meters (87.8 inches) of rain at the study location. There was no discharge recorded at the outlet in 79 percent of the events. During the full monitoring period 7635 m3 (2.69 x 105 ft3) of water entered at the inlet and 3982 m3 (1.40 x 105 ft3) left the system. Therefore, at least 48 percent of the incoming water never discharged from the 110th Cascade. However, the total impervious area draining to the system was carefully estimated to be approximately double that contributing to the inlet; therefore, the total runoff volume entering likely was also about double that measured at the monitored inlet. Given this, it would be more accurate to say that closer to 74 percent of the water entering the 110th Cascade was retained.

Table 6 presents estimated total pollutant mass loading reductions over the full monitoring period, both with and without the unmeasured inflows, with confidence limits for the latter set of estimates (Chapman and Horner 2010, NRC 2009). Including the estimated additional influent gives the most likely estimate of reductions. By either estimation technique, though, it is clear that the NW 110th Cascade attenuated the majority, or even the great majority, of the pollutant mass that would otherwise flow to Pipers Creek for most pollutants. This performance is chiefly a function of the great reduction of discharge volume. The low removal of dissolved phosphorus signifies the exception usually seen in vegetative treatment systems: vegetation decaying in the fall and winter appears to release in soluble form nutrients taken up in the growing season.

Table 7 gives the irreducible minimum and reliable maximum pollutant concentration values determined from the cascade discharge data, with volume-weighted average effluent concentrations for comparison (Chapman and Horner 2010). The reliable maximum is here defined as the event mean concentration (EMC) that was exceeded in only 10 percent of the runoff events, while the irreducible minimum is the EMC that was exceeded 90 percent of the time. These values were calculated as two-sided prediction intervals with $\alpha=0.10$, using non-parametric (ranks) methods (Helsel and Hirsch, 1991). Volume-weighted averages were computed by multiplying concentrations times flow volumes for each monitored storm, summing, and dividing by total volume. While the minimums and averages show that pollutant concentrations in the discharge would usually be substantially less, the relative certainty of not surpassing the maximum concentrations is 90 percent.

Table 6. Estimated Reductions in Pollutant Mass Loadings Over the Full Sampling Program at the 110th Cascade (Chapman and Horner 2010, NRC 2009)

Pollutant	Estimated Mass Loading Reduction Ignoring Unmeasured Inflows (%)	90% Confidence Limits on Estimate	Estimated Mass Loading Reduction Including Unmeasured Inflows (%)
Total suspended solids	84	72-91	93
Total nitrogen	63	53-74	82
Total phosphorus	63	49-74	83
Soluble reactive phosphorus	No significant decrease	-	28
Total copper	83	77-88	90
Total zinc	76	48-85	90
Total lead	90	84-94	93
Dissolved copper	67	50-78	79
Dissolved zinc	55	21-70	86
Dissolved lead	Usually not detected in inflow	-	Usually not detected in inflow
Total petroleum hydrocarbons (motor oil fraction)	92	86-97	96

Table 7. Effluent Pollutant Concentration Statistics for the NW 110th Street Cascade (Chapman and Horner 2010)

Pollutant	Irreducible Minimum	Reliable Maximum	Volume-Weighted Average
Total suspended solids (mg/L)	9	40	30
Total nitrogen (mg/L)	0.59	1.27	0.81
Total phosphorus (μg/L)	81	210	133
Soluble reactive phosphorus (μg/L)	22	86	36
Total copper (μg/L)	3.9	7.6	6.3
Total zinc (μg/L)	39	106	47
Total lead (μg/L)	1.6	6.6	4.5
Dissolved copper (μg/L)	1.5	4.7	2.9
Dissolved zinc (μg/L)	14	54	26
Dissolved lead (μg/L)	< 1	< 1	< 1
Total petroleum hydrocarbons (motor oil fraction, mg/L)	< 0.15	0.32	0.22

WDOE has set water quality standards for some of the pollutants in Table 7, particularly metals as dissolved quantities. Meeting the standards in the discharge could be a stormwater management objective. The metals standards are a function of water hardness, because the tendency of hardness producing minerals (mainly, calcium and magnesium) to reduce metal toxicity to aquatic life. Hardness tends to be relatively low in the Puget Sound area, making attainment of water quality standards sometimes difficult. The results for the NW 110th Street Cascade show that employing this strategy could meet the standards for zinc and lead under virtually all regional circumstances but would not necessarily meet the copper standard.

Bioretention Effectiveness and Relative Certainty

As pointed out earlier, performance comparisons among studies are difficult because of the many influencing variables, often undefined in reports, and always must be tempered accordingly. Bioretention facilities have been built and studied with and without impermeable liners and/or underdrains, either entirely eliminating (if lined) or reducing (if unlined but underdrained) infiltration. Obviously, this design feature would be expected to have a major influence on performance. Table 8 presents a comparison in surface runoff volume reduction with and without underdrains. Results for the unconstrained systems mirror those discussed above for Seattle's natural drainage systems. Installing an underdrain but leaving the facility unlined appears to cut the hydrologic advantage by roughly one-third to one-half, while adding a liner diminishes that advantage by around two-thirds. Therefore, these design features should only be incorporated for a good reason (e.g., high groundwater table; very restricted infiltration rate that cannot be sufficiently increased by soil amendment; buried contaminants in the soil below, which could be mobilized by concentrated infiltration). Without such a reason, bioretention cells and other ARCD practices should be built without these features for maximum infiltration opportunity and minimum surface discharge.

Table 8. Surface Runoff Volume Reduction Achieved by Bioretention Systems With and Without Underdrains (adapted from NRC [2009] and references cited)

Design	Location	Volume Reduction (%)	Reference
Unlined, no underdrain	Connecticut	99	Dietz and Clausen (2006)
Unlined, no underdrain	Pennsylvania	86	Ermilio and Traver (2006)
Unlined, no underdrain	Florida	98	Rushton (2002)
Unlined, no underdrain	Australia	73	Lloyd et al. (2002)
Unlined, with underdrain	Ontario	40	Van Seters et al. (2002)
Unlined, with underdrain	North Carolina	40-60	Smith and Hunt (2007)
Unlined, with underdrain	North Carolina	52-56	Hunt et al. (2008)
Unlined, with underdrain	Maryland	52-65	Davis et al. (2008)
Lined, with underdrain	North Carolina	20-29	Sharkey (2006)

The 20-29 percent of the inflow lost from the lined unit could only have departed via evapotranspiration. This result came from an installation in a location with the months of highest rainfall coinciding with the warmest months, maximizing evaporation, and the growing season, maximizing transpiration. Everything else being equal, this performance would not be expected in the Puget Sound region, with the highest rainfall in the coolest months outside the growing season. Nevertheless, design and, particularly, vegetation selection could increase ET in this climate; but research is necessary to determine that potential and how to optimize it.

Davis et al. (2009) summarized water quality performance registered by several bioretention studies in the eastern United States from New Hampshire to North Carolina. As shown in Table 9, the results were highly variable, most likely as a consequence of the many driving variables listed earlier. The results overall do not provide good indices of effectiveness and signify that relative certainty is quite poor. Along with the reports from the Seattle natural drainage systems, though, they do indicate the strong potential of bioretention to eliminate the discharge of almost all pollutant mass in the best applications and to meet or at least approach achievement of water quality standards at the point of release.

Table 9. Summary of Water Quality Performance of Eastern United States Bioretention Cells (after Davis et al. 2009, excluding laboratory and pilot tests)

Pollutant	Effluent Concentration Range	Mass Loading Reduction Range (%)
Total suspended solids	13-20 mg/L	54-99
Total nitrogen	0.80-4.38 mg/L	32-97
Total phosphorus	58-560 μg/L	negative 240-79
Total zinc	17-48 μg/L	54-99

Permeable Pavement Effectiveness and Relative Certainty

Permeable pavements include porous asphalt and concrete and various modular concrete and plastic grid products. All have received some testing, but results are reported in many different ways. The tests have generally shown substantial reductions of both runoff quantity and measures of pollutants. Apparently, mechanisms operate within the porous matrices to capture the various classes of pollutants relatively effectively. Already generally relatively low, remaining contaminants exiting from the pervious pavement would receive additional opportunity for capture in underlying soil before reaching groundwater.

St. John and Horner (1997) investigated the performance of a King County, WA porous asphalt road shoulder relative runoff quantity and quality from the traveled lanes and in comparison to a conventional asphalt and a gravel shoulder. Over a wet season the porous asphalt shoulder prevented 85 percent of the incident runoff from flowing farther on the surface. Samples of the road runoff and shoulder infiltrate exhibited the water quality characteristics in Table 10. The results are roughly equivalent to or better than those from the NW 110th Street Cascade, with a relatively high degree of certainty. Notably and in contrast to the cascade, soluble orthophosphate-phosphorus was clearly reduced to low concentrations in the permeable pavement.

Struck, Rowney, and Pechacek (2009) summarized case studies on various kinds of concrete and plastic grid systems. Concrete blocks over four types of base materials in Germany reduced copper in the infiltrate to 16-51 μg/L, zinc to 18-178 μg/L, and lead to < 4 μg/L. A concrete-block system in North Carolina reduced runoff volume by 66 percent and held total suspended solids in water passed through the blocks to 12.4 mg/L, total nitrogen to 0.98-2.77 mg/L, TP to 70-400 μg/L, and zinc to 8 μg/L. Brattebo and Booth (2003) examined two concrete and two plastic grids over five years. There was almost no runoff at any time from any gridded area. Copper in the infiltrate ranged from below detection to 1.3 μg/L and zinc from less than detection to 8.2 μg/L. Oil was never detected.

Table 10. Performance of a King County, WA Porous Asphalt Road Shoulder (after St. John and Horner 1997)

Pollutant	Road Runoff Concentrationa	Porous Asphalt Infiltrate Concentrationa	Mass Loading Reduction by Porous Asphalt (%)
Total suspended solids (mg/L)	140 ± 22	17 ± 4	97
Total phosphorus (μg/L)	358 ± 35	101 ± 15	94
Orthophosphate-phosphorus (μg/L)	19.1 ± 5.2	5.8 ± 1.0	90
Total copper (μg/L)	16.2 ± 2.4	4.8 ± 1.0	92
Total zinc (μg/L)	105 ± 15	39 ± 11	91
Total lead (μg/L)	39.7 ± 6.4	4.7 ± 0.8	97
Total petroleum hydrocarbons (motor oil fraction, mg/L)	2.5 ± 0.3	0.9 ± 0.1	95

a Given as mean ± 1 standard error

A coalition of public agencies and private professional associations and consultants has built an International Stormwater Best Management Practices Database[11]. The database is a reliable basis for characterizing effectiveness, because it incorporates only data collected using acceptable procedures and quality controls. Hence, this source is fully peer-reviewed. It is now primarily populated with conventional practices but will soon be supplemented with a range of ARCD methods. At this point, among those practices, only porous pavements are included. Table 11 summarizes the results from six studies on a variety of pavement types accepted into the database. Porous pavement technology requires further investigation on long-term sustainability, as well as stormwater management effectiveness; but the overall reports thus far are quite promising for greatly reducing runoff quantities and pollutant mass loadings and meeting or coming close to water quality standards.

Table 11. Statistics on Porous Pavement Infiltrate Water Quality from the International Stormwater Best Management Practices Database

Pollutant	Median	95% Confidence Limits
Total suspended solids (mg/L)	17	6-49
Total Kjeldahl nitrogena (mg/L)	1.23	0.44-3.44
Total phosphorus (µg/L)	90	50-150
Total copper (µg/L)	2.8	0.9-8.9
Total zinc (µg/L)	17	6-47
Total lead (µg/L)	7.9	1.6-38

a Ammonia plus organic nitrogen

Vegetated Roof Effectiveness and Relative Certainty

Dietz (2007) summarized retention without surface discharge of the precipitation falling on 10 vegetated roofs in Sweden, Michigan, North Carolina, and Oregon. The systems ranged from 2.0 to 12.7 cm in growth medium thickness and the roof slopes from 2.0 to 6.5 percent. Excluding one roof with a 2.0-cm medium, retention ranged from 58-71 percent. With infiltration not being a factor, ET had to be substantial. Because of the preponderance of precipitation in winter in the Pacific Northwest, it is generally thought that green roofs would not be very effective for stormwater management here. However, even the Portland, OR vegetated roof retained 69 percent of the rainfall (Dietz 2007). It also has the thickest medium, which might be one clue to boosting performance in this climate.

Pennsylvania State University has performed a large amount of green roof research (Berhage et al. 2009). The roofs tested retained over 50 percent of the total precipitation during the study period. During summer months nearly all the precipitation was retained. During the winter retention was smaller (< 20 percent). Seasonal effects appear to be a result of snow or freezing conditions; otherwise green roofs effectively retained up 0.4 inch (10 mm) of precipitation regardless of season. This result is encouraging for vegetated roof use in the Puget Sound region, where very cold conditions are much less frequent than in central Pennsylvania. The mean storm quantity in Seattle is 0.48 inch, meaning that the potential exists to achieve substantial retention of a fairly large number of storms. Water quality performance was not deemed good enough to discharge effluent without further treatment. This finding is discouraging to implementation of green roofs, since they are best suited to dense locations restricting use of other practices.

Benefits of Water Harvesting

To the extent that rain water can be harvested and directed to a use such as gray water supply or irrigation, the technique is 100 percent effective in reducing stormwater surface runoff and its contamination. Therefore, its effectiveness is expressed here in terms of water conservation potential. In downtown Seattle the King County Government Center collects enough roof runoff to supply over 60 percent of the toilet flushing and plant irrigation water requirements, saving approximately 1.4 million gallons of potable water per year (Puget Sound Action Team 2003). A much smaller public building in Seattle, the Carkeek Environmental Learning Center, drains roof runoff into a 3500-gallon cistern to supply toilets (Accetturo 2005).

ARCD costs

The USEPA (2007b) assembled a series of ARCD case studies, including costs. In general, the investigation concluded that:

... applying LID techniques can reduce project costs and improve environmental performance. In most cases, LID practices were shown to be both fiscally and environmentally beneficial to communities. In a few cases, LID project costs were higher than those for conventional stormwater management practices. However, in the vast majority of cases, significant savings were realized due to reduced costs for site grading and preparation, stormwater infrastructure, site paving, and landscaping. Total capital cost savings ranged from 15 to 80 percent when LID methods were used, with a few exceptions in which LID project costs were higher than conventional stormwater management costs (USEPA 2007b)

Among the Pacific Northwest case studies, Seattle's 2nd Avenue NW SEA Streets project saved $217,255 of the expected $868,803 cost (25 percent) of upgrading the street's previous "informal" drainage system to a conventional street curb-and-gutter configuration. Two parking lot rain garden retrofits in Bellingham saved 76 and 80 percent of the costs of the conventional stormwater management alternative of underground vaults. A design study for a Pierce County, WA subdivision using an integrated range of ARCD techniques estimated 20 percent savings compared to managing stormwater conventionally. On the other hand, in a design study for another subdivision in the same county maximizing ARCD opportunities, including home roof water collection, capital costs were estimated as about twice as high as for a conventional system. These costs were expected to be offset somewhat by operating savings over time. Portland's residential roof downspout disconnection program has cost the city $8.5 million thus far in materials and incentive payments but is expected to save $250 million in construction costs for piping to store an extra 1 billion gallons per year to prevent combined sewer overflows. Case studies in USEPA (2007b) for other areas around North America illustrate the general savings that can accrue from replacing conventional approaches with ARCD.

Conventional Stormwater Management Strategies

Stormwater management on the conventional level is very well developed, especially in Washington State. King County and WDOE were among the first jurisdictions in the nation to write comprehensive stormwater manuals. The same two entities took continuous simulation hydrologic modeling into the mainstream of the profession, ahead of just about everywhere else. However, the conventional approach has been found wanting, contributing to the strong role of stormwater in compromising resources. The NRC (2009)study concluded that site-by-site specification of controls on stormwater discharges, the usual practice in the conventional approach to stormwater management, is inadequate and should be replaced by integrated implementation of controls, designed as a system. In particular, the committee found that the prevailing practices have not served well to manage runoff from the most frequent, relatively small storm events; and practices that harvest, infiltrate, and evapotranspire stormwater would be superior. This section examines the capabilities and limitations of conventional practices, what role they can still play, and how they can be enhanced for better performance.

WDOE's (2005) Stormwater Management Manual for Western Washington provides a thorough catalog of the conventional practices at issue and can be consulted for full details about them. The practices under the infiltration and biofiltration categories can be recognized as identical in name to practices also in the ARCD category. The major differences as applied in that milieu versus conventionally come in the treatment of soils and vegetation. In ARCD applications soils are investigated for infiltration capability and amended if necessary to optimize it; whereas, conventionally, either the native soils are accepted as is or an infiltration BMP is rejected as a workable choice. In ARCD practices the vegetation palette generally has some diversity in more than one canopy layer, whereas conventional vegetated facilities tend more to be monocultures. In addition to the practices in Table 12, the manual has a volume (IV) of source controls, which have already been pointed out in the ARCD discussion.

Table 12. Conventional Stormwater Management Practices in Stormwater Management Manual for Western Washington (WDOE 2005)

Category	Specific Practices	Manual Reference[a]
Quantity control:		
Roof downspout controls	Downspout infiltration system	III-3.1.1
	Downspout dispersion system	III-3.1.2
	Perforated stub-out connection	III-3.1.3
Detention	Detention pond	III-3.2.1
	Detention tank	III-3.2.2
	Detention vault	III-3.2.3
Infiltration	Infiltration basin	III-3.3.10
	Infiltration trench	III-3.3.11
Quality control:		
Infiltration and bio-infiltration	Infiltration basin	V-7.4
	Infiltration trench	V-7.4
	Bio-infiltration swale	V-7.4
Sand filtration	Sand filter vault	V-8.8
	Linear sand filter	V-8.8
Biofiltration	Biofiltration swale	V-9.4
	Filter strip	V-9.4
Wet pool facilities	Wet ponds	V-10.3
	Wet vaults	V-10.3
	Treatment wetlands	V-10.3
Oil/water separators	Baffle-type	V-11.7
	Coalescing plate	V-11.7
Emerging (generally, commercial) technologies	Media filters	V-12.6.1-2
	Catch basin inserts	V-12.6.3
	Hydrodynamic separators	V-12.6.4
	High-efficiency sweeping	V-12.6.5

a Given as volume number (Roman numeral) and chapter section

Conventional Stormwater Management Strategies: Effectiveness and Relative Certainty in Water Quantity Control

As a consequence of the urban-induced runoff changes that cause flooding, erosion, and stream habitat damage, Puget Sound jurisdictions have long required some degree of stormwater runoff quantity mitigation for new developments. The most common approach has been to reduce flows through the use of detention ponds, which are intended to hold stormwater runoff from developed areas and release it at a slower rate than if undetained. Booth, Hartley, and Jackson (2002) reviewed the history and critiqued the effectiveness of the approach. The picture remains largely unchanged today, with the conventional practices described still prevailing and central to the approach of the Stormwater Management Manual for Western Washington (WDOE 2005).

Detention ponds can be designed to either of two levels of performance, depending on the desired balance between achieving downstream protection and the cost of providing that protection. A peak standard, the classic (and less costly) goal of detention facilities, seeks to maintain post-development peak discharges at their pre-development levels. Even if this goal is successfully achieved, the aggregate duration that such flows occupy the channel must increase, because the overall volume of runoff is greater. In contrast, a duration standard seeks to maintain the post-development duration of a wide range of peak discharges at pre-development levels. Yet, unless runoff is infiltrated, the total volume of runoff must still increase in the post-development condition. Thus, durations cannot be matched for all discharges because this excess water must also be released.

Applying these principles requires the use of some calculation procedure, a hydrologic model, to estimate pre- and post-development flows. Early protocols used the extremely simplistic "Rational Method," succeeded about 20 years ago by the "Curve Number" method originally introduced by the Soil Conservation Service of the U.S. Department of Agriculture. Several flaws, resulting in detention ponds that did not meet desired performance criteria, were soon recognized in this method: (1) ponds were assumed to be empty at the beginning of storms, a condition often not the case in the Puget Sound winter climate; (2) the model commonly overestimated pre-development flows, giving the wrong targets for post-development design; (3) the method was still based on a peak standard, ignoring problems associated with increased flow durations.

To counter these problems King County and WDOE introduced continuous simulation hydrologic models based on the HSPF (Hydrologic Simulation Program-Fortran) model, respectively, KCRTS (King County Runoff Time Series) and WWHM (Western Washington Hydrologic Model). These and other jurisdictions also converted to duration control standards, intended to match pre- and post-development flow durations for all discharges above a chosen threshold. From the standpoint of developers these changes were controversial, because they led to substantially larger detention ponds, consuming more land and costing more than before.

From the environmental protection standpoint, the use of a threshold (on development size and, hence, runoff production) ignores cumulative effects of numerous sub-threshold actions summing to a considerable hydrologic alteration. Booth and Jackson (1997) had earlier discovered that one-quarter of impervious area added to King County watersheds from 1987 to 1992 fell below the threshold. Horner et al. (2002) assessed various aspects of water quantity and quality control BMP application in four King County watersheds and found that only 12-31 percent of the developed area was served by any quantity control practices. This dearth appeared to be associated with vesting under old regulations and some development predating any regulations, in addition to the threshold.

Booth and Jackson (1997) performed an analysis to determine how much detention volume would be required to prevent the urban stream channel damage, which they and others had demonstrated, based on a duration-based standard based and KCRTS modeling and assuming no infiltration. They concluded that effective runoff mitigation in the Pacific Northwest requires pond volumes of 3 to as much as 14 cm-ha per ha of developed land (0.10-0.46 acre-ft/acre). With associated berms, control structures, and maintenance access roads, such a facility could occupy more than 10 percent of the total area of a development. Ponds of that size have never been built, and probably never will for economic and political reasons. If half of the runoff production could be avoided by ARCD mechanisms, ponds could shrink to around the sizes being built under current standards and still protect streams. The possibility certainly exists to achieve greater attenuation in the contributing catchment, to apply ARCD-type soils amendment and vegetation to the pond to increase infiltration and ET, or both.

Conventional Stormwater Management Strategies: Effectiveness and Relative Certainty in Water Quality Control

The optimal stormwater management practice can provide needed quantity control, substantially reduce pollutant mass loadings, and produce an effluent concentration within acceptable limits, as gauged by water quality standards in the receiving water. The effectiveness of conventional infiltrative practices in providing any or all of these benefits depends on the extent of infiltration that occurs, just as with their ARCD counterparts. Basin-type conventional water quality control practices can be designed to provide detention for quantity control. Ground-based practices that drain fully, such as detention ponds, biofilters, and media filters constructed in earth, generally have some incidental infiltration, although it is not usually accounted for in design. If that incidental infiltration is considerable, mass loading reduction will benefit both from volume decrease and pollutant capture in the device. Otherwise, cumulative mass of contaminants will not be reduced much or at all over what the pollutant capture mechanisms provide. Wet ponds and treatment wetlands hold water because they do not infiltrate much, a condition developing through soil structural changes with saturation if not the case at construction. Of course, any practice built in a hard structure will not infiltrate at all.

For practices not designed for infiltration, their effectiveness in reducing pollutant concentrations depends on a variety of pollutant removal mechanisms, the chief one being filtration and settling of suspended solids, which captures any other contaminants associated with the particles. The longer the residence time in the device, the more sedimentation will occur, because of the inverse relationship of settling velocity and particle size. Mechanisms removing dissolved pollutants (e.g., adsorption, ion exchange, precipitation) also depend on time to function effectively. Conventional BMPs are usually designed to treat runoff from the relatively frequent, small storms (e.g., 6-month frequency, 24-hour duration) and pass larger flows through rapidly or bypass them. The grounds for this practice are that these storms convey the great majority of the pollutant loadings, and targeting bigger storms requires increasingly larger treatment systems for diminishing benefits. The effectiveness reports here are a function of this design philosophy.

The International Stormwater Best Management Practices Database, introduced above, is the best basis for characterizing conventional BMP effluent quality in terms of pollutant concentrations. For supporting material on intertational stormwater BMP, see Appendix 4F. In an exercise to compare conventional to ARCD treatment in lowering pollutant concentrations, this author compared the medians and 95 percent confidence intervals in Table F1 with the volume-weighted averages, irreducible minimums, and reliable maximums in Table 7 summarizing the Seattle NW 110th Street Cascade's performance, which is fairly typical in relation to other ARCD data cited above. These various statistics are, of course, not strictly comparable but do provide similar indicators of effectiveness and relative certainty. Table 13 shows the comparison in terms of when the conventional BMP concentrations were generally "higher," "comparable," or "lower" in relation to those in the natural drainage system effluent. There is no statistically quantitative basis underlying or implied in these ratings, simply a general overlap or deviation in one direction or another. Dissolved lead is not included because the influent to the cascade was generally below detection, differing from any entry in the database.

The results provide a convenient means of comparing the conventional BMPs to one another and to a typical ARCD installation. Wet ponds and treatment wetlands are quite comparable to one another and the natural drainage system cascade in exhibiting the highest effluent quality. Somewhat less effective overall are media filters and conventional biofilters, with detention ponds and hydrodynamic devices showing the lowest performance.

While this analysis indicated that it is possible to produce effluents with conventional practices of comparable quality to ARCD alternatives, the two comparable conventional types are essentially non-infiltrative. While they would provide an uncertain amount of evapotranspiration, they are unlikely to be comparable in mass loading reduction to a system that extracts the great majority of the surface runoff. Most media filters and all hydrodynamic devices use hard structural containments and offer no infiltration or transpiration and little evaporation.

Table 13. Comparison of Effluent Water Quality from Conventional Stormwater BMPs and Seattle NW 110th Street Cascade Natural Drainage System[a]

Pollutant[b]	Detention Ponds	Wet Ponds	Treatment Wetlands	Biofilters	Media Filters	Hydrodyn. Devices
TSS	C	L	L	C	L	H
T N	H	H	H	C	C	H
T P	H	L	C	H	C	H
D P	H	H	H	H	H	H
T Cu	H	C	C	H	H	H
T Zn	H	L	L	L	L	H
T Pb	H	C	C	H	C	H
D Cu	H	H	No data	H	H	H
D Zn	C	C	No data	C	H	H
% of cases C or L	22	67	71	44	56	0

[a] H—measures of central tendency and dispersion generally higher in conventional BMP than cascade effluent; L— measures of central tendency and dispersion generally lower in conventional BMP than cascade effluent; C—measures of central tendency and dispersion generally comparable

[b] TSS—total suspended solids, T—total, N—nitrogen, P—phosphorus, D—dissolved, Cu—copper, Zn—zinc, Pb—lead, Cd—cadmium

The California Department of Transportation (Caltrans 2004, Barrett 2005) performed an extensive study of conventional BMPs for highway applications. It was discovered that extended-detention ponds and biofiltration swales and filter strips infiltrated 30-50 percent of the influent, depending on soils and storm characteristics, giving an unanticipated boost to mass loading reduction, the statistical ranges of which are shown in Table 14. It should be noted that these facilities were not designed to provide for water quantity control, nor were they evaluated for that function.

These data can be compared with the mass loading performance of the NW 110th Street Cascade as shown in Table 6. All BMPs but the hydrodynamic device were fairly effective in cutting mass emissions of TSS and particulate metals. The wet pond and sand filters were at least the equals of the cascade in this regard. It should be noted that the Caltrans detention pond was designed for a 3-day holding time for the target storm, the longest generally used for this device; and it performed better than often reported elsewhere. The advantage of the greater flow volume reduction afforded by the cascade showed up more with respect to the dissolved metals and, especially, the nutrients, for which the cascade was estimated to remove 82-83 percent of the total nitrogen and phosphorus. Nevertheless, the generally better than expected performance of the Caltrans BMPs shows the way on how the most can be gained from conventional BMPs, a subject discussed below.

Table 14. Ranges of Percentage Mass Loading Reductions by Caltrans (2004) BMPs[a]

Pollutant[b]	Detention Pond	Wet Pond	Biofiltration Swales	Biofiltration Filter Strips	Sand Filters	Hydrodyn. Device
TSS	70-80	85-95	70-80	80-90	85-95	20-30
TKN	35-45	Neg.-30	45-55	10-45	25-60	Neg.-25
Part. P	65-80	Neg.-45	30-75	Neg.-50	Neg.-80	15-65
D P	0-30	Neg.	Neg.	Neg.	10-20	10-20
Part. Cu	85-95	95-100	85-95	90-95	95-100	Neg.-25
Part. Zn	80-85	95-100	85-95	85-95	90-95	25-50
Part. Pb	75-85	95-100	80-85	90-95	80-95	45-85
D Cu	25-35	35-70	55-65	70-80	10-55	Neg.-30
D Zn	45-55	65-80	70-80	65-80	65-95	Neg.-30
D Pb	60-70	65-85	55-75	80-85	60-90	0-10

[a] Expressed as 90 percent confidence limits of percentage reductions from inlet to outlet rounded to nearest 5 percent; there were different numbers of BMPs in each category, and two different designs of sand filters; neg.—negative.

[b] TSS—total suspended solids, TKN—total Kjeldahl nitrogen (ammonia plus organic nitrogen), Part.—particulate (total minus dissolved), P—phosphorus, D—dissolved, Cu—copper, Zn—zinc, Pb—lead, Cd—cadmium

Potential Advances in Conventional Practices

The often limited or nonexistent infiltration and evapotranspiration occurring in non-infiltrative conventional stormwater practices limits their ability to achieve effective control over runoff quantity and pollutant mass loadings, even if they can be designed and built to attain relatively high contaminant concentration reductions. While sand and other media filters are often constructed with concrete vaults, they can also be established in earth or without a hard bottom. Indeed, Austin, TX, which pioneered one type of stormwater sand filter, promotes such a design

(City of Austin 1988) and has many such open-bedded filters. There are no technical limitations to amending soils to promote infiltration, a technique institutionalized in ARCD practice, in otherwise conventional detention basins, biofiltration swales and filter strips, and open-bedded media filters. Likewise, vegetation could be converted from the often monocultural (usually, grass) stand to more diverse forms in several canopy layers in detention ponds and biofiltration swales and filter strips. Such plantings are thought to give a boost to water storage, infiltration and ET. Treatment wetlands already often have such diversity, but the fringe of wet ponds could be planted in this way too.

Advances in Industrial Stormwater Treatment

As discussed above under the topic Special Considerations for Industrial Land Use, industries have source control and other ARCD options but will still sometimes have to treat runoff to meet water quality objectives. There have been recent advances in technology for these applications, documented in a peer-reviewed study growing out of a Puget Sound-area challenge by environmental groups to the general stormwater permit for boatyards. In settlement the contending parties and their technical representatives designed a study to determine the effectiveness of three treatment technologies in removing total suspended solids (TSS) and total and dissolved copper, lead, and zinc from boatyard runoff. Taylor Associates, Inc. (2008) conducted the study under contract to the parties, who managed it and reviewed and approved its results (Box 9).

Box 9. Technologies investigated for the Boatyard stormwater treatment technology study.

- StormwateRx® Aquip™—an enhanced filtration device consisting of a pretreatment chamber followed by a series of inert media that filter particulates and adsorb dissolved substances;
- Siemens Water Technologies, Inc. Wastewater Ion Exchange System—a device consisting of an activated carbon chamber to remove organics followed by tanks containing ion exchange resins to remove specific ionic contaminants; and
- Water Techtonics, Inc. Wave Ionics™ Electro-Coagulation System—a device applying electric current to coagulate particles so that they either sediment, if more dense than water, or rise to the top of the water column, if buoyant, for capture.

Table 15 presents results for the first two technologies in Box 9, omitting those for the third, which was much less effective. Mass loading reductions would be similar to the concentration reductions in the table; because inflow and outflow quantities were essentially the same without infiltration and very little ET. The ion exchange unit was the more effective of the two treatments, especially in capturing zinc. Performance for copper was similar, but neither technology would guarantee meeting the water quality standard for that metal in Puget Sound or most of its freshwater tributaries at the discharge. After completion of the study StormwateRx® reached an agreement with Siemens to market their two systems together as a “treatment train” to gain the advantage of zinc capture and, probably, also somewhat reduce copper.

Table 15. Results of Investigation of Enhanced Filtration and Ion Exchange for Industrial Stormwater Treatment (after Taylor Associates, Inc. 2008)

Pollutant[a]	Technology[b]	Average Concentration Reduction[c] (%)	Irreducible Minimum Concentration	Reliable Maximum Concentration
TSS	Enhanced. filt.	84	< 1	2
	Ion exch.	95	< 1	3
Total copper	Enhanced filt.	95	4.0	21.0
	Ion exch.	99	2.0	19.4
Total zinc	Enhanced filt.	60	46	153
	Ion exch.	97	6	31
Total lead	Enhanced filt.	68	< 1	< 1
	Ion exch.	97	< 1	< 1
Dissolved copper	Enhanced filt.	93	2.9	18.3
	Ion exch.	99	2.0	17.4
Dissolved zinc	Enhanced filt.	58	43	138
	Ion exch.	97	5	29
Dissolved lead	Enhanced filt.	ND[d]	ND[d]	ND[d]
	Ion exch.	76	< 1	< 1

[a] TSS—total suspended solids; all concentration units µg/L, except TSS (mg/L)
[b] Enhanced filt.—enhanced filtration (Aquip); ion exch.—ion exchange (Siemens)
[c] Reported for composite samples (one grab sample also collected early in each storm)
[d] ND—not detected in influent and thus could not calculate

Strategies for Ubiquitous, Bioaccumulative, and/or Persistent Pollutants (BPT)
Certain toxicants found in stormwater are very widespread (ubiquitous) and persist in the same or related toxic forms over an extended period in the environment. In some cases these contaminants concentrate in the tissues of living organisms (bioaccumulation). In others they persist because of being in chemical elemental form (e.g., metals), and hence are not degradable, or are organic but degrade slowly. Some ubiquitous, persistent pollutants are relatively soluble (e.g., copper, zinc) and are, consequently, difficult to remove from runoff by conventional or even advanced treatment techniques to a level protective of aquatic life.

Box 10. BPT substances in stormwater identified by the NRC (2009) report.

- Coal tar-based asphalt sealants, a common source of polycyclic aromatic hydrocarbons (PAHs), a group including carcinogens, mutagens, and otherwise toxicants;
- Creosote- and chromated copper arsenate (CCA)-treated wood;
- Zinc in tires, roof shingles, and downspouts;
- Copper in brake pads and boat hull antifouling paints;
- Various heavy metals in fertilizers; and
- Road deicers, principally sodium chloride.

The NRC (2009) report pointed out that potentially less harmful substitutes exist or could likely be developed for many of these products, and also that federal legislation exists under with USEPA could restrict or ban them. Generally, this action is not happening, although the committee cited the bans on leaded gasoline and the pesticide diazanon as leading to documented large decreases in the environment. In the absence of federal action, some jurisdictions are taking action on the local level. For example, Austin, TX and Dane County, WI have banned coal tar-based asphalt sealants. These actions suggest possible strategies for the Puget Sound region to consider in advancing the product-substitution source control under a broad ARCD program. Washington got a strong start in implementing this strategy in March 2010 by becoming the first state to phase out and eventually ban copper and other metal toxicants in brake pads, pending the governor's signing the legislation[11].

These measures would address the threat of acute and chronic toxicity effects on aquatic organisms from metal and organic pollutants. They would contribute to Results Chain strategies RC6 (Stormwater) C2 generally and RC 7 (Wastewater) C1, specifically C1(2) (support Persistent Bioaccumulative Toxic program implementation)(Neuman et al. 2009).

Construction Site Stormwater Management

Land cleared of vegetation and not otherwise stabilized yields much more sediment compared to the original area well covered with plants and to the same area restablized with vegetative cover following construction. Both measurements and estimates using a mathematical model (Revised Universal Soil Loss Equation) indicate 30 to more than 1000 times as much soil loss after compared to before clearing (Novotny and Chesters 1981). Sediment discharge to receiving water bodies presents numerous threats, the delineation of which is beyond the scope of this chapter.

Effective controls are availableto prevent erosion and sediment movement and cut soil loss to a very small fraction of the maximum potential. WDOE's (2005) Volume II is a thorough compendium of those practices. However, these techniques are often not applied effectively. The NRC (2009) committee diagnosed the problem, at least in part, as a failure to recognize the most effective practices and apply them first if appropriate to the construction site's circumstances. To address this problem the committee outlined a recommended approach that puts the numerous types of practices in a hierarchy (Box 11). The first priorities are practices that avoid erosion, followed by those that do not entirely prevent it but limit it greatly. Sediment trapping practices are the lowest priority, because they are not nearly as effective as the erosion prevention and limiting options, although they still should be considered as backups where risk of damaging sediment release still exists.

These improvements to construction site stormwater management would likely addressthreats to salmon spawning and rearing habitat, aquatic food webs, and water quality in all downstream waters, including Puget Sound arising from the negative effects of eroded sediments and toxicants from construction materials, processes, and wastes. They would also contribute to Results Chain identified by Neumann et al. (2009).

Box 11 Recommended Construction Site Stormwater Control Measures (after NRC (2009))

1. As the top priority, emphasize construction management practices as follows:

- Maintain existing vegetation cover, if it exists, as long as possible.
- Perform ground-disturbing work in the season with smaller risk of erosion, and work off disturbed ground in the higher risk season.
- Limit ground disturbance to the amount that can be effectively controlled in the event of rain.
- Use natural depressions and plan excavation to drain runoff internally and isolate areas of potential sediment and other pollutant generation from draining off the site, so long as safe in large storms.
- Schedule and coordinate rough grading, finish grading, and final site stabilization to be completed in the shortest possible time overall and with the shortest possible lag between these work activities.

2. Stabilize with cover appropriate to site conditions, season, and future work plans. For example:

- Rapidly stabilize disturbed areas that could drain off the site, and that will not be worked again, with permanent vegetation supplemented with highly effective temporary erosion controls until achievement of at least 90 percent vegetative soil cover.
- Rapidly stabilize disturbed areas that could drain off the site, and that will not be worked again for more than three days, with highly effective temporary erosion controls.
- If at least 0.1 inch of rain is predicted with a probability of 40 percent or more, before rain falls stabilize or isolate disturbed areas that could drain off the site, and that are being actively worked or will be within three days, with measures that will prevent or minimize transport of sediment off the property.

3. As backup for cases where all of the above measures are used to the maximum extent possible but sediments still could be released from the site, consider the need for sediment collection systems including, but not limited to, conventional settling ponds and advanced sediment collection devices such as polymer-assisted sedimentation and advanced sand filtration.

4. Specify emergency stabilization and/or runoff collection (e.g., using temporary depressions) procedures for areas of active work when rain is forecast.

5. If runoff can enter storm drains, use a perimeter control strategy as backup where some soil exposure will still occur, even with the best possible erosion control (above measures) or when there is discharge to a sensitive waterbody.

6. Specify flow control SCMs to prevent or minimize to the extent possible:

- Flow of relatively clean off-site water over bare soil or potentially contaminated areas;
- Flow of relatively clean intercepted groundwater over bare soil or potentially contaminated areas;
- High velocities of flow over relatively steep and/or long slopes, in excess of what erosion control coverings can withstand; and

- Erosion of channels by concentrated flows, by using channel lining, velocity control, or both.

7. Specify stabilization of construction entrance and exit areas, provision of a nearby tire and chassis wash for dirty vehicles leaving the site with a wash water sediment trap, and a sweeping plan.

8. Specify construction road stabilization.

9. Specify wind erosion control.

10. Prevent contact between rainfall or runoff and potentially polluting construction materials, processes, wastes, and vehicle and equipment fluids by such measures as enclosures, covers, and containments, as well as berming to direct runoff.

Strategies for management of bacteria in stormwater

The following are the conclusions from a review by (Horner and Osborne 2005) that is available as supporting information to this update (Appendix 4G).

Two general methods exist to prevent or reduce shellfish bed contamination by urban stormwater: pollution source controls and runoff treatment. Source controls separate the points of pollution origin from contact with rainfall or runoff; if the separation is complete, they are 100 percent effective in preventing contamination. Runoff treatments attempt to remove pollutants already in runoff; they can reduce but cannot entirely prevent contamination, unless all runoff infiltrates the soil and only emerges to surface water after full pathogen die-off.
The literature review investigated commonly used urban stormwater treatment techniques: constructed wetlands, ponds, media filters, vegetated filter strips and swales, and hydrodynamic devices. It also covered the small amount of information available on stormwater disinfection. Excluding disinfection, constructed wetlands yielded the best performance in terms of fecal coliform reduction efficiency and effluent quality. All other options reviewed, except disinfection, generally produced effluents with FC concentrations two to three orders of magnitude higher than the presumed target of ~101/100 mL. Ultraviolet disinfection has been shown, as would be expected, to lower concentrations below detection. However, it is the most logistically difficult and expensive option.
Even with constructed wetlands, effluent FC concentrations were still generally an order of magnitude above the ~101/100 mL target. The major exception to this observation was the StormTreat system, a modular, manufactured constructed wetland on the commercial market, which reduced influent concentrations ranging 102-104/100 mL to a mean below detection. Kadlec and Knight (1996), in evaluating results from municipal wastewater treatment in wetlands, offered an important clue regarding why the StormTreat system can out-perform large, more naturalistic constructed wetlands in FC reduction. They concluded that constructed wetland outflow concentrations cannot consistently be reduced to near zero, or even close, without disinfection, if the wetland is open to wildlife. This point was also illustrated in the research of Grant et al. (2001) on the man-made Talbert Marsh, concluding that the seagull droppings were a direct source of FCs emerging from the marsh to the surf zone along Huntington Beach, CA. The

StormTreat units are not conducive to wildlife occupancy or access by domestic animals. The Caltrans (2004) experience with a constructed wetland in an urban freeway right of way adds evidence supporting this conclusion. This wetland was not easily accessible or attractive to wildlife and domestic animals. It exhibited the lowest bacterial effluent concentrations among the installations reviewed, although they were still considerably above the StormTreat levels. The StormTreat system thus would deserve serious further consideration for application in the Puget Sound region from the performance standpoint.

Climate Change Relative to Stormwater Management

Rosenberg et al. (2009) assessed the impact of climate change on Puget Sound's stormwater infrastructure with predicted precipitation distributions and the Hydrologic Simulation Program—FORTRAN (HSPF) to simulate stream flow in two urban watersheds. They found that the range of precipitation projections is too large to predict effects on engineering design, and actual changes could be hard to distinguish from natural variability. Nonetheless, they suspected that the rainfall records of the past will not be a reliable design basis. As reported earlier, a shift toward higher cool-season and lower warm-season storm runoff is expected for the Puget Sound region. This pattern would tend to necessitate enlarging stormwater management facilities, but the extent of that need is not clear at this point.

Synthesis of stormwater management strategies

A leading theme of the NRC (2009) committee, and this segment of Chapter 4-2, is a comprehensive approach to stormwater management. A manifestation of that approach, now formalized in some stormwater programs around the nation, is a five-factor framework (DeBarry 2004) built around management of:

- A groundwater recharge volume;
- A water quality volume;
- A channel protection storage volume;
- An overbank flood protection peak flow rate; and
- An extreme flood protection peak flow rate.

WDOE's (2005) approach implicitly incorporates much of the philosophy but lacks a groundwater recharge element and treats channel protection in terms of duration instead of volume explicitly. Requirements are hence not set here for recharge and are needed to fill out the region's strategy.

DeBarry (2004) noted that the final two factors are easily managed using the traditional post- to pre-development peak rate match for the large, infrequent storms at issue. However, the first three, involving volume management, require a more innovative strategy, in particular one using ARCD methods. The strategies emerging from the assessment presented in this segment of the chapter can be approached from the five-factor model. Essentially, they boil down to making every attempt to meet the three volume (or duration) targets by selecting, as appropriate to the location being managed, ARCD practices from among those summarized in Table 5. The intention is to apply practices in a decentralized (i.e., close to the source), integrated fashion. If a full, scientifically based analysis shows that it is indeed impossible to meet the targets with these practices, then one can turn to in lieu fees, trading credits, and/or conventional techniques to make up the difference. It was pointed out earlier that advances can be brought to bear on those conventional practices to raise their effectiveness, in part by adopting ARCD elements like soil amendment and more diverse planting. Data presented in this chapter (e.g., from the 2nd Avenue NW SEA Streets project) showed that it may even be possible in some instances to contribute strongly to meeting overbank and extreme flood protection requirements with these strategies.

Three general key strategies arise from the review of ARCD and conventional stormwater management practices and the special topics:

Strategy 1: As the principal basis of urban stormwater management, apply Aquatic Resources Conservation Design practices in a decentralized (i.e., close to the source), integrated fashion to new developments, redevelopments, and as retrofits in existing developments as necessary to meet established protection and restoration objectives. If a full, scientifically based analysis shows that it is indeed impossible to meet objectives with these practices, employ, first, in lieu fees or trading credits or, as a second priority option, conventional stormwater management practices according to Strategy 2.

Strategy 2: Employ conventional stormwater management practices when Strategy 1 options do not fully meet objectives. Increase the effectiveness of conventional vegetation- and soil-based practices whenever possible by using ARCD landscaping techniques. Apply enhanced filtration, ion exchange, or a treatment train involving both in industrial situations when source controls and ARCD measures are insufficient to meet objectives.

Strategy 3: Address special stormwater problems as follows:

3A. Promote source control under a broad ARCD program by assessing ubiquitous, bioaccumulative, and/or persistent pollutants that can only be controlled well by substituting with non-polluting products and enact bans on the use of products containing those pollutants.
3B. Improve construction site stormwater control by prioritizing, first, construction management practices that prevent erosion and other construction pollutant problems; second, practices that minimize erosion; and, last, sediment collection after erosion has occurred.
3C. To counteract dispersed sources of pathogens that compromise shellfish production and other beneficial uses, implement strong source controls and treat remaining sources with subsurface-flow constructed wetlands, assuming additional research and development verifies the promise of that technique.

Domestic Wastewater Issues and Strategies

Introduction

With municipal wastewater treatment plants now converted to a secondary level or treatment, the former problems associated with biodegradable organics in discharges are largely solved for collected wastewater (Box 12).

Box 12. Issues concerning wastewater treatment addressed by WDOE (2008)

- *Combined sewer overflow*—discharge directly into a receiving water without treatment from a wastewater collection system designed to carry sanitary sewage and stormwater in a single pipe to a treatment facility, resulting from precipitation causing a high storm runoff quantity exceeding the plant's capacity;
- *Sanitary sewer overflow*—discharge onto the land surface or a water body when the capacity of a separate sanitary sewer is exceeded, normally during storm events due to unplanned inflow from the surface and infiltration from the subsurface;
- *Advanced municipal wastewater treatment*—for constituents other than the solids, biodegradable organics, and pathogens, for which secondary treatment plants are designed, remaining as threats, principally nitrogen in the Puget Sound region but could also include phosphorus and toxic metals in some cases and may expand to include pharmaceuticals and other organic consumer products now emerging as concerns; and
- *On-site wastewater treatment*— the general ineffectiveness of the conventional septic tank and drain field in preventing delivery of nitrogen and, to a lesser extent, phosphorus and pathogens to nearby receiving waters via subsurface flow.

WDOE (2008) addressed each of those issues in a document intended to establish a consistent basis for the design and review of plans and specifications for sewage treatment works. Each subject represents a specialized technical field with many complexities and references of varying level available to address them.

Combined Sewer Overflows

Because of state actions under federal mandates, combined sewer overflow (CSO) interdiction programs have been underway in the Puget Sound region for a number of years, but remain incomplete. WDOE (2008) covers numerous techniques to address CSOs, prominently including: 1) Institutional controls (e.g., sewer use ordinances, pollutant source pretreatment programs); 2) Source controls (e.g., ARCD water quantity and quality controls, conventional stormwater quantity and quality controls, construction site controls, catch basin cleaning); 3) Collection system controls (e.g., sewer separation, infiltration and inflow control, maximizing use of existing system, valves and other flow regulating devices, flow diversion); 4) Storage technologies(e.g., in-line storage, off-line near-surface storage. deep tunnel storage); 5) Centralized treatment technologies (e.g., use of excess primary treatment capacity during storms to obtain some treatment, addition of primary or secondary capacity); 6) On-site treatment (e.g., off-line near-surface storage and sedimentation, screening, vortex technologies, disinfection, dissolved air floatation, filtration).

There are many considerations in selecting among the profuse alternatives, with cost being a leading one. With the documented cost savings usually accruing to ARCD methods relative to highly structural ones, there is growing interest in applying these techniques as retrofits in cities with combined sewers. The Center for Low Impact Development engaged in a two-phase project under Water Environment Research Foundation sponsorship to identify strategies for implementing decentralized ARCD controls for urban retrofits in general and CSO reduction in particular. Phase 1 of the project (Weinstein et al. 2006) demonstrated the technical feasibility of the concept by drawing on the experience of a number of early adopters using decentralized controls to complement their existing municipal stormwater and wastewater infrastructure.

However, institutional and programmatic issues required further study to broaden the use of a distributed, decentralized stormwater approach.

The second phase of the project (Weinstein et al. 2009) evaluated implementation strategies for incorporating decentralized controls into an infrastructure management system. The distributed nature and multiple environmental benefits of decentralized controls necessitate an integrated and inter-departmental management approach. The Phase 2 report emphasizes policy and financing strategies, along with guidance for using common stormwater models to analyze decentralized controls. Case studies and programmatic and regulatory examples detail alternatives to expedite the adoption of decentralized controls. This work can be put to work in the Puget Sound region through the following key strategy:

Key Strategy: Bolster incomplete combined sewer overflow reduction programs by using ARCD techniques identified for application in that setting to decrease stormwater flows.

Sanitary Sewer Overflows

Sanitary sewer overflows (SSO) are considered unauthorized discharges not covered by National Pollutant Discharge Elimination System (NPDES) permits, and must be reported as spills (WDOE 2008). SSOs are mostly a function of the condition of sewer lines and their ability to exclude infiltrating subsurface water at transition points in the system (e.g., pipe joints). While it can be expensive, the strategy for addressing this problem where it exists is relatively straightforward: trace the sources of excess water and repair leaks.

Advanced Municipal Wastewater Treatment

Background

Starting with the removal of treatment plant effluents from Lake Washington about 50 years ago, the outfalls of all Puget Sound-area municipally operated plants discharging to fresh waters were moved to salt water. Over the past 30 years any plants with primary treatment (solids settling) only have been upgraded to a secondary level (adding biological decomposition of organics). These improvements have greatly reduced the problems associated with discharging relatively high solids and organics. However, secondary treatment leaves high concentrations of nitrogen (N) and phosphorus (P) present in the influent domestic wastewater and, if there are contributory sources, may also discharge metals and complex organic chemicals not well decomposed in the process.

Nitrogen has become a particular concern in Puget Sound with the recognition of its role in reducing oxygen and stressing or killing fish. Secondary treatment reduces biochemical oxygen demand (BOD) from influent concentrations greatly but still leaves as much as 30 mg/L in the effluent. Material composing BOD also biodegrades and consumes oxygen. While N in the organic and ammonia or ammonium forms exerts an oxygen demand, the principal oxygen-depletion mechanism is through eutrophication, which is algal-growth-promoted, mainly by N in nitrate form, which is readily taken up and used by algae. The usual initial by-product of organic decomposition in the normally circumneutral pH of natural water is ammonium ion, with the ammonia form, toxic to aquatic life, very suppressed except at high pH. The ammonium converts, in the bacterial-mediated process called nitrification, first to nitrite-N and then, quickly in well aerated waters, to nitrate-N.

Treatment processes can enhance nitrification and make it more complete. Such processes must be aerobic, in the presence of oxygen. Once nitrate forms, though, it can only be further converted in a process mediated by anaerobic bacteria, capable of living only without oxygen, termed denitrification, where the end product is nitrogen gas. Full conversion of nitrogen from wastewater to an innocuous substance can only be performed through the sequence of nitrification-denitrification, thus requiring a sequence of opposite oxygen environments.

The growth of marine algae is generally limited by an insufficient supply of N in relation to available carbon and P. In other words, if that deficiency should be relieved by the inflow of nitrogen in wastewater, the limitation is relaxed, more primary production occurs, and algal biomass builds up in the eutrophication syndrome. If favorable growth conditions change (e.g., temperature and light decrease with the onset of autumn), cells die in large numbers and are decomposed by aerobic bacteria, taking dissolved oxygen from the water. The high P content of wastewater can reinforce eutrophication, by supplying P should it become limiting in the presence of very abundant N. P is also the limiting nutrient in fresh waters more often than N. While municipal treatment plants are no longer a concern in the eutrophication of Puget Sound's fresh waters, on-site treatment systems and small packaged treatment plants are.

Municipal treatment plants impose pretreatment requirements on industrial dischargers to limit the influent heavy metals, which are toxic to aquatic life in varying degrees. While these plants generally have no particular processes designed to remove metals, particulate settling and incorporation in sludge reduce their concentrations. Municipal plants remain sources, but metals enter receiving waters in other important pathways, particularly via stormwater runoff and atmospheric deposition.

Domestic wastewater also contains a host of chemicals present in pharmaceuticals, cosmetics, cleaning products, industrial materials, etc. that are variably removed in secondary treatment. These chemicals are just emerging as concerns and not much is known yet about their quantities, environmental dynamics, and effects on organisms in the receiving water.

Advanced Municipal Treatment Options and Their Effectiveness and Relative Certainty

Advanced wastewater treatment, often termed tertiary treatment, can be accomplished by a number of technologies, which can be combined in different ways depending on treatment objectives, wastewater characteristics, plant configuration, and costs. This review concentrates on well developed methods that can potentially address major threats to Puget Sound and advance PSP Action Agenda and Results Chain strategies. We report primarily on nitrogen (N) and to a lesser extent, phosphorous (P) and heavy metals. While the most advanced technologies generally address all classes of pollutants, specific study of their effectiveness in removing emerging chemicals from waste streams is very sparse at this point.

According to the perhaps most authoritative textbook in the field, Metcalf and Eddy (2003), there are 12 recognized classes of physical and chemical processes for the removal of the general range of residual contaminants in treated wastewater effluents. There are also various biological techniques to reduce N and P, which are complicated by the alternating aerobic-anaerobic environments that must be produced for the initial nitrification step followed by denitrification. There are many permutations of the treatment system in both suspended- and attached-growth forms. The system can be set up in separate chambers, although it is possible for the processes to proceed in the same tank with different oxygen environments in different parts of it (Metcalf and Eddy 2003). Filtration can be added to improve nitrogen removal over what is possible with nitrification-denitrification. Depending on the process selection and operation, total N in the effluent can be reduced to a concentration of 3-10 mg/L (Metcalf and Eddy 2003).

The typical biological P removal system has an anaerobic reactor ahead of an activated-sludge aeration tank, with activated sludge recycling from the secondary clarifier to the head of the process and, in some designs, an intermediate reduced-oxygen chamber (Metcalf and Eddy 2003). Anaerobic organisms in the first vessel accumulate complex forms of P and release simplified, more directly usable forms like orthophosphate, which are incorporated into cell tissue in the aerobic reactor and subsequently settled out. A total P effluent concentration of $\leq$ 2 mg/L can be attained through biological treatment alone (Metcalf and Eddy, Inc. 2003). In a survey of 23 advanced municipal wastewater treatment plants nationwide, USEPA (2007c) found that a concentration as low as 0.3 mg/L was often attained. The same survey established that addition of aluminum- or iron-based coagulants to wastewater followed by tertiary filtration can reduce total P concentrations in the final effluent to near or below 0.01 mg/L. Combined systems can be designed to treat for both P and N.

Among the 12 classes of available physical and chemical treatment alternatives, membrane technologies represent the best combination of a relatively high state of development and treatment versatility, including removal of both N and P (Metcalf and Eddy 2003). The primary membrane applications and their abbreviations and filter pore sizes as designated by the Water Environmental Federation (WEF 2006) are: (1) low-pressure membranes- microfiltration (MF) and ultrafiltration (UF), (2) nanofiltration (NF), and (3) reverse osmosis (RO). A membrane bioreactor (MBR) is a combination of suspended-growth activated sludge secondary biological treatment with MF or UF replacing the conventional secondary clarifier, either submerged in the bioreactor or placed in a subsequent unit. The arrangement can precede discharge or serve as pretreatment for highly advanced NF or RO follow up (WEF 2006).

Table 16 gives a membrane technology performance summary for the contaminants of most concern in the Puget Sound ecosystem. MF following conventional secondary treatment does not improve overall performance as much as the MBR configuration. In comparison to the purely biological treatments covered above, capable of achieving total N and P concentrations of 3-10 and 0.3-2 mg/L, respectively, MBR is comparable or a slight improvement for N and somewhat better for P. Adding RO to MF or UF conveys major performance advantages at increased cost. Based on the data presented above, coagulant addition and filtration can improve P removal even more, although without nearly as much advantage for N reduction.

Table 16. Effectiveness of Membrane Technology Tertiary Treatments in Comparison to Conventional Activated Sludge Secondary Treatment (after WEF 2006, Metcalf and Eddy 2008)[a]

Pollutant	Conventional Activated Sludge	Conventional Activated Sludge + MF	MBR	Conventional Activated Sludge + MF or UF + RO
Biochemical oxygen demand	5 – 30	< 2 – 10	< 2 – < 5	< 2 – < 5
Ammonia-nitrogen	15 – 25	20 – 35	< 1	≤ 0.1
Nitrate-nitrogen	1 – 2	20 – 35	< 3 – < 10[c]	ND – < 2
Total nitrogen	15 – 35	5 – 30[b]	< 3 – < 10[c]	≤ 0.1
Total phosphorus	1 – 10	0.1 – 8[d]	< 0.2 – 1[d]	≤ 0.5

[a] MF—microfiltration; MBR—membrane bioreactor; UF—ultrafiltration; RO—reverse osmosis; ND—not detectable
[b] Total Kjeldahl nitrogen (nitrogen in organic plus ammonia or ammonium ion forms)
[c] < 3 mg/L with pre- and post-anoxic zones; < 10 mg/L with pre-anoxic zone only
[d] With chemical addition

The reverse osmosis system is also effective in metals removal. Reported effluent concentrations (all as total recoverable metals in µg/L) are: arsenic—< 2 – 5, cadmium—< 1 – < 10, chromium—< 10 – < 50, and mercury—< 0.2 – < 2 (WEF 2006). These concentrations would be sufficiently low to meet or approach fresh and marine water receiving water quality standards at the discharge point (i.e., without dilution).

Synthesis of Strategies

If nitrogen discharge from a municipal treatment plant is a serious threat, reverse osmosis tertiary treatment with highly efficient filtration as a pretreatment is the most effective and certain solution (Metcalf and Eddy 2008, WEF 2006). The same solution can apply to phosphorus and toxic metals. However, if phosphorus alone is the problem, then coagulation and tertiary filtration appears to offer an equivalent or possibly even better solution. That latter situation is not likely to occur in the Puget Sound region though.

Key Strategy: If nitrogen discharge from a municipal treatment plant must be reduced below 1 mg total nitrogen/L to remove a threat to marine dissolved oxygen resources, apply reverse osmosis tertiary treatment with highly efficient filtration as a pretreatment. If analysis demonstrates that a lesser reduction will suffice, apply membrane bioreactor treatment.

The nitrogen-reduction strategy supports Results Chain strategies RC 7 (Wastewater) C1, specifically C1(8) and C1(9) (remediation actions to address low dissolved oxygen); and C3, specifically C3(1) (advanced wastewater treatment), C3.1.1 (improved nitrogen removal at wastewater treatment plants), and C3.4 (technologies that reduce nutrients) (Neuman et al. 2009).

On-site Wastewater Treatment

On-site wastewater treatment refers to systems treating effluents, most often domestic, from a single building or a small cluster. A typical conventional on-site treatment system consists of a septic tank and a soil absorption field (drain field). The septic tank functions as an anaerobic bioreactor promoting partial digestion of organic matter and solids settlement. The drain field distributes septic tank effluent through perforated pipes into the soil for additional biological processes, adsorption, and filtration before infiltration of the water to groundwater.

These systems work well if they are installed in areas with appropriate soils, hydraulic capacities, and separation from groundwater; designed and installed properly; maintained in good operating condition; and replaced when necessary to maintain performance. These criteria are often not met, however. Only about one-third of the United States land area has soils suitable for conventional on-site systems (USEPA 2002). In addition, septic tanks and drain fields are frequently not large enough for the flows from modern houses, system densities sometimes exceed the capacity of even suitable soils to effectively process waste, and installations are too close to ground or surface waters. As a consequence failure rates are known to be high but are difficult to establish precisely for a number of reasons, including varying definitions of failure. The failure rate in Washington is estimated at 33 percent (USEPA 2002).

The main consequences of failure are contamination of groundwater, surface waters, or both by nitrates, phosphorus, and/or disease-causing bacteria and viruses. Table 17 summarizes the performance capabilities of a well-functioning conventional drain field. It is evident in the table that the conventional system can produce a BOD as low as or even lower than secondary treatment.

Table 17. Reductions of Problematic Pollutants by a Conventional Drain Field in Good Working Order (after USEPA 2002, Jantrania and Gross 2006)

Pollutant[a]	Typical Septic Tank Effluent Concentration	Soil Removal Efficiency (%)	Concentration Remaining in Water Exiting Drain Field
BOD (mg/L)	130-150	90-98	2.6-15
Total N (mg/L)	45-55	10-40	27-50
Total P (mg/L)	8-12	85-95	0.4-1.8
Fecal coliforms (CFU/100 mL)	106-107	99-99.99	103-106

[a] BOD—biochemical oxygen demand; N—nitrogen, P—phosphorus, CFU—colony forming units.

In contrast to P, soils do not have a similar capacity for N, which is mostly in the highly soluble ammonium form in a septic tank effluent. Nitrification is rapid in the aerobic environment of the soil, with the result that nitrate, also highly soluble and at a very elevated concentration, makes up most of the total N moving out from the drain field (USEPA 2002). The nitrate can penetrate to groundwater, where it can be a health risk if the water is drawn for potable supply. Methemoglobinemia, ("blue baby syndrome") is the most well-established effect, but others are suspected (Washington State Department of Health 2005). As Table 17 shows, conventional systems are highly efficient in reducing disease-causing organisms, represented by fecal coliforms, an indicator organism present with numerous pathogens in sewage. However, the numbers are so high that even strong removal can leave counts in the many thousands for each 100 mL of water. Failed systems would do far worse yet. As with nitrogen, waters are threatened by pathogen-contaminated groundwater and surfacing effluent, especially from installations near the shore.

Advanced On-site Treatment Options and Their Effectiveness and Relative Certainty

Advanced treatment categories and specific examples put forth by Jantrania and Gross (2006) include aerobic treatment units, media filters using such substances as sand, peat foam and textiles, natural systems such as treatment wetlands and greenhouses, waterless toilets and disinfecting systems using UV light or chlorination.

Aerobic treatment units are essentially miniature versions of devices commonly used in municipal secondary treatment plants. Numerous packaged units are on the commercial market for small-scale applications. Given their process similarity to larger scale secondary treatment, their effluent quality is also similar: approximately 15-35 and 1-10 mg/L total N and P, respectively. Since they normally discharge to soil, additional reductions are possible there. However, since the reason to use such an option is often poor soils, further capture may not be much. Pathogens are not greatly reduced by secondary treatment, or the miniaturized aerobic treatment units, alone and require disinfection if soil or other conditions make pathogens a threat to receiving waters.

Media filters are normally placed between a septic tank and drain field. The most common are sand filters in single-pass or recirculating form. Packaged units with sand and other media are available on the market. Table 18 gives reported performance data (USEPA 2002).

Table 18. Pollutant Concentrations in On-site Scale Media Filter Effluents from Nine Studies Reported in the Literature (after USEPA 2002)

Pollutant	Single-Pass Filters	Recirculating Filters
Biochemical oxygen demand (mg/L)	2-4	3-10
Total nitrogen (mg/L)	28-38	16-32
Fecal coliforms (No./100 mL)	102-103	101-104

There is a great deal of literature on treatment wetlands for municipal wastewater treatment, generally in relative small communities. These reports can give further insight on what might be possible to achieve in on-site treatment wetlands. The summary data (USEPA 2000) show that only systems with open water can achieve much nitrogen reduction, to < 10 mg total Kjeldahl N/L. With a long hydraulic retention time (up to 15 days), effluent can be maintained at < 1.5 mg total P/L. As discussed above, pathogen reduction is expected to be better with a submerged-bed wetland (without a free water surface), but this configuration would be disadvantageous for nitrogen removal. Nevertheless small treatment wetlands appear to offer promise of effluent quality roughly comparable to discharges from aerobic treatment units and media filters (USEPA 2000).

The Washington State Department of Health (WSDOH 2005) reviewed specific on-site nitrogen-reducing technologies for WDOE. The review concluded that biological nitrification-denitrification is the only process that has been demonstrated to be technically and economically feasible for on-site applications. The process must be structured to manage the alternating aerobic/anaerobic environments required for the two steps. The aerobic phase can be accomplished by a variety of aerobic treatment units or media filters. The anoxic phase requires addition of organic carbon to nourish the bacteria. One option is to recycle nitrified wastewater back through the septic tank, where the anaerobic, high carbon environment can facilitate denitrification. Another is to provide a separate denitrification chamber and external carbon source.

Numerous non-proprietary (public domain) and proprietary (patented) systems exist to provide these functions. USEPA and the National Sanitation Foundation have collaborated on an Environmental Technology Evaluation protocol to test and verify the performance of these systems (WSDOH 2005). Six technologies have completed the testing and exhibited total N effluent concentrations in the range 14-19 mg/L. An additional nine products have been tested in USEPA demonstration projects and reported a wide range of 2-83 mg total N/L in effluents. The NITREX™ system, a processed wood fiber media filter[13], showed the best performance, discharging 2.0-2.4 mg/L, but was tested at only two installations (WSDOH 2005).

Synthesis of Strategies for Controlling Wastewater

The reported results show that a specialized solution must be sought if nitrogen is the threat to be countered. A reasonable strategy for the Puget Sound region would be to test further the available system(s) exhibiting the best results in limited assessments. It would not be appropriate to adopt any generic type of system, as different versions of a general technology type have exhibited varying performance. It would also not be appropriate to adopt a promising system that has not been thoroughly tested under regionally prevailing conditions.

If pathogens are a threat, it is highly likely that no system designed for nitrogen removal will reduce them sufficiently; and disinfection will be necessary. Since small-scale disinfection is not very well developed, additional research and development work will be necessary in this area.

Phosphorus is a threat to lakes with heavily inhabited shorelines using conventional on-site systems. The review did not reveal as much work to address P with advanced treatments as appeared for N. The limited results available do not indicate alternatives that can lower concentrations to the levels that affected lakes would need for substantial water quality improvement. Lake eutrophication from on-site systems is a localized problem in comparison to the more broadly distributed threats from nitrogen and pathogens discharged to marine waters with oxygen depletion and shellfish bed contamination.

As an alternative to the continuing on-site treatment, with presently developed advanced treatment options of only limited effectiveness relative to the treatment need to meet environmental objectives, would be to construct sewers and a municipal treatment plant where on-site systems are a leading threat. However, it would be essential to apply this strategy in such a way that it did not lead to additional development, the storm runoff from which could undo progress made from eliminating on-site wastewater discharges.

Key Strategy: If discharges from on-site wastewater treatment systems are a serious threat to: (1) marine dissolved oxygen resources as a result of nitrogen; or (2) shellfish production or contact recreation as a result of pathogens, assess as possible solutions: (1) construct sewers and a municipal treatment plant, with advanced treatment for nitrogen if that is the threat, to replace problem on-site systems; or (2) apply advanced on-site treatment, tested and verified to reduce the problem sufficiently to remove the threat (note: at this point more testing is required for both on-site nitrogen removal systems and small-scale disinfection).

Strategies to Manage Agricultural Activities for Water Quality Protection

Best Management Practice Guidance

Best management practices are available to serve virtually every agricultural function. The Natural Resources Conservation Service (NRCS) of the U.S. Department of Agriculture has codified them in the National Handbook of Conservation Practices (NHCP, NRCS 2007b), containing more than 165 practices. For each BMP the NHCP presents a standard and a conservation practice physical effects (CPPE) worksheet. The conservation practice standard contains information on why and where the practice is applied and sets forth the minimum quality criteria that must be met during its application for it to achieve its intended purpose(s). The CPPE worksheet provides guidance on how the application of the practice will affect the resources (soil, water, air, plants, animals and human) and the concerns associated with each of those resources. It reflects the best estimate of the effects, either positive or negative, of the practice on the resource concerns. For many practices there is also a conservation practice information sheet and a job sheet. The information sheet contains a photograph of the installed practice, a definition or description, where it is commonly used, and a brief, qualitative description of its conservation effects when it is properly applied. The job sheet provides detailed guidance on the application of the practice and has worksheets that can be used to document the practice plan and design for a specific site. The NHCP cautions that the standards themselves should not be used to plan, design, or install a conservation practice; instead the specific analogous standard developed by the state in which the agricultural site is located should be consulted to insure that all state and local criteria are met. The Washington State Department of Agriculture's website[14] provides some related guidance, but the department apparently has not comprehensively revised the NHCP practices.

Mostaghimi et al. (2001) summarized 18 commonly used practices from the NHCP and two other emerging ones (integrated pest management and precision farming). Each BMP is described and assessed for its impact on the physical, chemical, and biological processes that control the generation and transport of pollutants. Each practice is classified in two ways: (1) purpose (source reduction, transport interruption, or a combination), and (2) mechanism (managerial, structural). Next, each account covers the situations and pollutants for which the practice is appropriate. A third section discusses any negative effects and limitations. Finally, the presentation suggests combinations of practices that can synergize effectiveness. This book chapter is a useful adjunct to the NHCP.

Special Considerations for Nutrient Management Pertinent to Puget Sound

As pointed out earlier, N is generally the nutrient limiting, and therefore controlling, algal growth in marine waters; while P usually plays that role in fresh waters. However, relieving a limitation with excess supply of one nutrient can switch control to the other and stimulate algal growth further. This eutrophication process yielding high algal production results in a number of problems in the affected water. Oxygen depletion caused by the death and decay of marine algae stimulated by nitrogen supply is the issue of greatest prominence now in the Puget Sound region. Relative to the interplay between these two nutrients, some important considerations in selecting and applying agricultural BMPs have emerged in the research literature.

Based on up to 30 years of experimental and monitoring data from a Pennsylvania watershed, Pionke et al. (2000) found that most of the surface runoff and P export originated from areas near the stream. About 90 percent of the form of P most available to algae exported in outflow was generated during the largest seven annual storms. In contrast, nearly all N was exported in the nitrate form and originated as subsurface flow entering the soil or groundwater some distance from the stream. These flows occurred during non-storm flow periods. Heathwaite et al. (2000) estimated the primary P-yielding zone to constitute < 20 percent of the total contributing area, while the upland N-generating areas were around 60 percent, in locations of well-draining soils and high fertilizer and manure application. The researchers concluded that strategies for managing P should focus on the few larger storms and relatively small critical source areas (Heathwaite et al. 2000). Conversely, strategies for N control depend more on balancing nitrogen application over the watershed. Without integrating strategies, solving one water quality problem can aggravate another. For example, practices applied to reduce surface runoff and P export by increasing infiltration will typically increase groundwater recharge and nitrate leaching.

Sharpley et al. (2001) took observed that the small areas disproportionately exporting phosphorus are located where high soil P, or P application in mineral fertilizer or manure, coincide with high runoff or erosion potential. They argued that the overall goal of efforts to reduce P loss to water should involve balancing P inputs and outputs at farm and watershed levels by optimizing animal feed rations and land application of P as mineral fertilizer and manure, targeted to relatively small but critical watershed areas for P export. These authors elaborated on the need to manage N and P together, citing more examples of how practices directed toward one can enlarge a problem with the other. For example, basing manure application on crop N requirements to minimize nitrate leaching to groundwater can increase soil

P and its export. In contrast, reducing surface runoff losses of total P via conservation tillage can enhance N leaching and even increase algal-available P transport (Sharpley et al. 2001)

Sharpley et al. (2001) also advocated development of a technically sound framework that recognizes critical sources of P and N export so that optimal strategies at farm and watersheds scales can be implemented to manage both together in the best way. One approach is to employ a phosphorus index to target its management toward critical P-source areas and apply N-based management on all other areas. As reported by Sharpley et al. (2003), the P indexing approach has been adopted by 47 states. The index ranks site vulnerability to P loss by accounting for source (soil test P, fertilizer, and manure management) and transport factors (erosion, runoff, leaching, and connectivity to a stream channel). Some states have modified the index to reflect local conditions and policies. Careful consideration must be given to the potential long-term consequences of N management on P loss and vice versa.

Lowrance et al. (1984) provided early support, but also qualification, on the value of a riparian buffer between agricultural fields and streams. They studied a subwatershed of the Little River, Georgia, 1568 ha (3872 acres) in area, with 30 percent riparian forest; 41 percent row crops; 13 percent pasture; and 16 percent roads, residences, fallow land, and other uses. They estimated nitrogen and phosphorus retention by the riparian buffer at 68 and 30 percent, respectively, of the inputs. Soils of the riparian ecosystem presented ideal conditions for denitrification: high organic matter from input of forest litter; seasonal waterlogging leading to anaerobiasis; and large inputs of nitrate-N in subsurface flow. Denitrification outputs alone were enough to remove all of the N inputs from upland fields to the riparian zone. The lack of an analogous process limited P retention.

The results from Lowrance et al. (1984) point out the particular importance of tributary riparian buffers to interrupt nitrogen transport to N-limited marine waters. However, the findings regarding P export originating mainly near streams indicate that riparian buffers can also play a larger role in stemming discharge of that nutrient than indicated by the modest 30 percent retention, not from interrupting transport but from excluding agricultural operations where they have the greatest potential to yield P to the receiving water (Lowrance et al. 1984).

Strategy Effectiveness for Nutrient Management and Relative Certainty

The NRCS *National Handbook of Conservation Practices* (NHCP) gives qualitative indications of practice effectiveness but not the quantitative data needed for objective comparisons among options. USEPA's (2003c) National Management Measures for the Control of Nonpoint Pollution from Agriculture partially fills this gap, drawing on the extensive but uncoordinated research on the performance of some of the many NHCP practices. The USEPA document covers BMPs for nutrient management, pesticide management, erosion and sediment control, animal feeding operations, grazing management, and irrigation water management.

Synthesis of Strategies

NRCS's NHCP is an exhaustive compendium of practices available to prevent or reduce contamination of water, and the USEPA (2003c) manual is one source of quantitative effectiveness and relative certainty data. Relative to the particular concern with eutrophication, the research literature offers a clear and conclusive strategy for integrated management of nitrogen and phosphorus sources.

Key Strategy: Upgrade the implementation of established agricultural best management practices, especially where agricultural runoff is: (1) a eutrophication threat as a result of nitrogen (N) and/or phosphorus (P); or (2) a threat to shellfish production or contact recreation as a result of pathogens. Manage nitrogen and phosphorus in concert by: (1) employing a phosphorus index to target management of critical P source areas, generally near receiving waters; and (2) applying N-based management to all other areas. Maintenance of riparian buffers advances both facets of the strategy by keeping agricultural activities out of the potentially most critical P production area and providing a sink for N to capture the majority of it before it can enter the water.

Further work is needed to institutionalize this strategy in watersheds subject to the negative impacts of eutrophication and, in general, to provide more directed guidance on the full range of contaminant issues to Puget Sound agricultural concerns.

Forestry Water Pollution Sources and Control Strategies

The potential for sediment delivery to streams is a long-term concern from almost all forestry harvesting activities and from forest roads regardless of their level of use or age (i.e., for the life of the road). Other pollutants, generally of somewhat shorter concern, include nutrients, increased temperature, toxic chemicals and metals, organic matter, pathogens, herbicides, and pesticides (USEPA 2005). Forest harvesting can also affect the hydrology of a watershed, with potential to degrade aquatic ecosystems. Forestry activities can also affect the aquatic habitats through physical disturbances caused by construction of stream crossings, equipment use within stream corridors, and placement of slash or other debris generated by forestry activities within streams. Negative impacts and conditions vary with location and water body type, but in general the ecological conditions that management measures and BMPs are intended to protect include the following (USEPA 2005) (Box 11):

Box 11. Attributes of watersheds that best management practices put forth by the USEPA (2005) are intended to protect.

- General water quality, by minimizing inputs of polluted runoff;
- Water temperature, by ensuring an adequate (but not excessive) and appropriate amount of shade along shorelines and stream banks;
- Nutrient balance, by providing for an adequate influx of carbon and nutrients that serve as the basis of aquatic food chains;
- Habitat diversity, by ensuring that inputs of large organic debris to the aquatic system are appropriate for the system; and
- Hydrologic processes, by limiting disturbances to stream flow patterns, both seasonal and annual.

As with the segment on agriculture, we present only a brief summary of strategies available to reduce water pollution from forestry activities. Again, numerous practices have been developed and well institutionalized to control the full range of activities. This review generally describes the system existing in Washington and sources of best management practice information. With no attempt at comprehensiveness here, further detailing could be a follow up in a future edition of the Puget Sound Science Update.

This account addresses PSP Results Chain strategies RC6 (Stormwater) C2, specifically C2.8 (private stewardship and incentives for pollution prevention). Forestry activities are both private and public, under the jurisdictions of the U.S. Forest Service and the Washington Department of Natural Resources.

Strategies to Manage Forestry Activities for Water Quality Protection: Best Management Practice Guidance

Modern management of forestry in relation to water resources in Washington stems from 1986, when Tribes, the timber industry, the state, and the environmental community decided to try to resolve contentious forest practices problems through negotiations as an alternative to competitive lobbying and court cases. This process resulted in the first Timber Fish Wildlife (TFW) agreement in 1987.

Over the years of TFW operation, regulation and management of forestry for the protection of water resources became well developed in Washington. Forest Practices Rules [15] , a compilation of 15 chapters of the Washington Administrative Code (WAC), establish standards for forest practices such as timber harvest, pre-commercial thinning, road construction, fertilization, and chemical application (Title 222 WAC). They give direction on how to implement the Forest Practices Act (chapter 76.09 Revised Code of Washington [RCW]) and Stewardship of Non-industrial Forests and Woodlands (chapter 76.13 RCW). The rules are designed to protect public resources such as water quality and fish habitat while maintaining a viable timber industry. They are under constant review through the adaptive management program. The Washington Department of Natural Resources' (WDNR) Forest Practices Board Manual[16] is an advisory technical supplement to the forest practices rules. It consists of 26 sections containing the BMPs for the full slate of forestry activities and detailed guidance for their proper implementation.

Listing of certain species of Pacific salmon as endangered or threatened resulted in a new round of TFW activity in the late 1990s. The interagency caucus formed issued the Forests and Fish Report[17] to the Forest Practices Board and the Governor's Salmon Recovery Office presenting recommendations for the development and implementation of rules, statutes, and programs for the protection and recovery of salmon. The general goal was to develop biologically sound and economically practical solutions to protect and improve riparian habitat on non-federal forest lands in the state, known as the "forestry module" for Washington's Statewide Salmon Recovery Strategy. This report does not outline BMPs per se but has influenced their development and adoption.

Strategy Effectiveness and Relative Certainty

The Forest Practices Board Manual does not provide data on the effectiveness and relative certainty of the BMPs covered. However, substantial performance data have been compiled in federal and state reports. USEPA's (2005) National Management Measures to Control Nonpoint Source Pollution from Forestry gives such data for many practices associated with pre-harvest planning, streamside management, road construction and subsequent management, harvesting, forest regeneration, fire management, revegetation, chemical application, and wetlands management. The TFW Cooperative Monitoring, Evaluation, and Research Committee sponsored three performance studies during the 1990s. Rashin and Grabin (1992) assessed riparian management zone regulations for protection of stream temperature. Rashin and Grabin (1993) covered BMPs for aerial application of forest pesticides. Finally, Rashin et al. (1999) reported on the performance of forest road and timber harvest practices. These works provide a basis for the following key strategy.

Key strategy: Upgrade the implementation of established forestry best management practices to protect stream water quality and hydrology in the vicinity of forestry activities and minimize the delivery of pollutants from those activities to downstream receiving waters, including Puget Sound.

Footnotes:

[1] http://www.epa.gov/owow/watershed/whatis.html

[2] http://water.usgs.gov/GIS/huc_name.html#Region17

[3] http://www.merriam-webster.com/dictionary

[4] http://water.washington.edu/Outreach/FactSheets/lwd.pdf

[5] http://www.ecy.wa.gov/programs/sea/wetlands/wfap/

[6] Stormwater control measures, also known as best management practices (BMPs)

[7] This account is adapted from NRC (2009) and was originally written for that report of the author of this section

[8] http://www.epa.gov/nps/lid/#guide

[9] http://www.lowimpactdevelopment.org/publications.htm#LID_National_Manuals

[10]

http://www.seattle.gov/util/About_SPU/Drainage_&_Sewer_System/GreenStormwaterInfrastructure/NaturalDrainageProjects/index.htm

[11] http://www.bmpdatabase.org/

[12] http://daily.sightline.org/daily_score/archive/2010/03/09/wa-approves-first-copper-brake-pad-ban

[13] http://www.deschutes.org/deq/nitrex.htm

[14] http://agr.wa.gov

[15] http://www.dnr.wa.gov/BusinessPermits/Topics/ForestPracticesRules/Pages/fp_rules.aspx

[16] http://www.dnr.wa.gov/BusinessPermits/Topics/ForestPracticesRules/Pages/fp_board_manual.aspx

[17] http://www.dnr.wa.gov/Publications/fp_rules_forestsandfish.pdf

References

Acceturro, A. 2005. Seattle Highlights Rainwater Harvesting at ARCSA 2005.

HarvestH20.com, available online athttp://www.harvesth2o.com/seattle.shtml.

Azous, A.L. and R.R. Horner (eds.). 2001. Wetlands and Urbanization: Implications for the Future. Lewis Publishers, Boca Raton, FL.

Baker, D.B., R.P. Richards, T.T. Loftus, and J.W. Kramer. 2004. A New Flashiness Index: Characteristics and Applications to Midwestern Rivers and Streams. Journal of the American Water Resources Association 40(2):503-522.

Barrett, M.E. 2005. Performance Comparison of Structural Stormwater Best Management Practices. Water Environment Research 77(1):78-86.

Battin, J., M.W. Wiley, M.H. Ruckelshaus, R.N. Palmer, E. Korb, K.K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences 104(16): 6720–6725.

Beechie, T.J., and T.H. Sibley. 1997. Relationship Between Channel Characteristics, LWD, and Fish Habitat in NW Washington Streams. Transactions of the American Fisheries Society 126:217–229.

Berghage, R.D., D. Beattie, A.R. Jarrett, C. Thuring, F. Razaei, andT.P. O'Connor. 2009. Green Roofs for Stormwater Runoff Control. U.S. Environmental Protection Agency, National Risk Management Research Laboratory, Cincinnati, OH.

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G. M. Kondolf, P. S. Lake, R. Lave, J. L. Meyer, T. K. O'Donnell, L. Pagano,

B. Powell, and E. Sudduth. 2005. Synthesizing U.S. RiverRestoration Efforts. Science 308:636-637.

Bolton, S, D.B. Booth, and D.R. Montgomery. 2003. Restoration of Puget Sound Rivers: Do We Know How to Do It? In Montgomery, D.R., S. Bolton, D.B. Booth, and L. Wall (eds.), Restoration of Puget Sound Rivers, University of Washington Press, Seattle, WA, pp. 483-490.
Booth, D.B. 1990. Stream-Channel Incision Following Drainage-Basin Urbanization. Water Resources Bulletin 26:407-417.

Booth, D.B. 1991. Urbanization and the Natural Drainage system—Impacts, Solutions, and Prognoses. The Northwest Environmental Journal 7:93-118.

Booth, D.B., D. Hartley, and R. Jackson. 2002. Forest Cover, Impervious-Surface Area, and the Mitigation of Stormwater Impacts. Journal American Water Resources Association 38(3):835-845.

Booth, D.B. and C.R. Jackson, 1997. Urbanization of Aquatic Systems – Degradation Thresholds, Stormwater Detention, and the Limits of Mitigation.Water Resources Bulletin 33(5):1077-1090.

Booth, D.B. and L. Reinelt. 1993. Consequences of Urbanization on Natural Systems—Measured Effects, Degradation Thresholds, and Corrective Strategies. Proceedings of the Watersheds ‘93 Conference, U. S. Government Printing Office, Washington, D.C.

Booth, D.B., J.R. Karr, S. Schauman, C.P. Konrad, S.A. Morley, M.G. Larson, P.C. Henshaw, E.J. Nelson, and S.J. Burges. 2001. Urban Stream Rehabilitation in the Pacific Northwest. Final Report, EPA Grant # R82-5284-010, University of Washington, Seattle, WA.

Brabec, E.A.2009. Imperviousness and Land-Use Policy: Toward an Effective Approach to Watershed Planning. Journal of HydrologicEngineering 14(4):425-433.

Brattebo, B.O., and D.B. Booth. 2003. Long-Term Stormwater Quantity and Quality Performance of Permeable Pavement Systems. Water Research 37(18):4369-4376.

Brierley, G.J. and K.A. Fryirs (eds.). 2008. River Futures: An Integrative Scientific Approach to River Repair. Island Press, Washington, DC.

Brinson, M.M. 1993. A Hydrogeomorphic Classification for Wetlands.U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Brookes, A. and F.D. Shields, Jr. (eds.). 1996. River Channel Restoration: Guiding Principles for Sustainable Projects. John Wiley and Sons, Chichester, U.K.

Brooks, K.N., P.F. Ffolliott, H.M. Gregersen, and L.F. DeBano. 2003. Hydrology and the Management of Watersheds, 3rd Ed. Iowa State University Press, Ames, IA.

Brown, K. 2000. Urban Stream Restoration Practices: An Initial Assessment. Prepared for USEPA by Center for Watershed Protection, Ellicott City, MD

Cairns, J., Jr. 1988. Increasing Diversity by Restoring Damaged Ecosystems. In E.O. Wilson (ed.), Biodiversity, National Academies Press, Washington, DC, pp. 333-343.

California Department of Transportation (Caltrans). 2004. BMP Retrofit Pilot Program Final Report. California Department of Transportation, Sacramento, CA.

Center for Watershed Protection. 1998. Rapid Watershed Planning Handbook: A Comprehensive Guide for Managing Urbanizing Watersheds. Center for Watershed Protection, Ellicott City, MD.

Chapman, C. 2006. Performance Monitoring of an Urban Stormwater Treatment System.

M.S.C.E. thesis, Department of Civil and Environmental Engineering, University of Washington, Seattle, WA.

Chapman, C. and R.R. Horner. 2010. Performance Assessment of a Street-Drainage Bioretention System. Water Environment Research 82(2):109-119.

Chin, N.T. 1996. Watershed Urbanization Effects on Palustrine Wetlands: A Study of the Hydrologic, Vegetative, and Amphibian Community Response Over Eight Years. M.S.C.E. thesis, Department of Civil and Environmental Engineering, University of Washington, Seattle, WA.

City of Austin. 1988. Environmental Criteria Manual. Watershed Protection Department, City of Austin, Tex.

Clay, C.H. 1995. Design of Fishways and Other Fish Facilities. CRC Press, Boca Raton, FL.

Climate Impacts Group. 2009. The Washington Climate Change Impacts Assessment, Elsner, M.M., J. Littell, and L.W. Binder (eds.). Center for Science in the Earth System, Joint Institute for Study of the Atmosphere and Oceans, University of Washington, Seattle, WA.

Cooper, C. 1996. Hydrologic Effects of Urbanization on Puget Sound Lowland Streams.Masters Thesis, Department of Civil and Environmental Engineering, University of Washington, Seattle, WA.

Cooke, G.D., E.B. Welch, S.A. Peterson, and S.A. Nichols. 2005. Restoration and Management of Lakes and Reservoirs, 3rd Ed. Taylor and Francis, Boca Raton, FL.

Cooke, S.S. and A.L. Azous. 2001. Characterization of Central Puget Sound Basin Palustrine Wetland Vegetation. In Azous, A.L. and R.R. Horner (eds.), Wetlands and Urbanization: Implications for the Future, Lewis Publishers, Boca Raton, FL, pp. 69-95.

Darby, S. and D. Sear. 2008. River Restoration: Managing the Uncertainty in Restoring Physical Habitat. John Wiley and Sons, Chichester, U.K.

Davenport, T.E. 2003. The Watershed Project Management Guide. Lewis Publishers, Boca Raton, FL.

Davis, A.P., W.F. Hunt, R.G. Traver, and M. Clar. 2009. Bioretention Technology: An Overview of Current Practice and Future Needs. Journal of Environmental Engineering 135(3):109-117.

DeBarry, P.A. 2004. Watersheds: Processes, Assessment, and Management. John Wiley and Sons, Inc., Hoboken, NJ.

de Waal, L.C., A.R.G. Large, and P.M. Wade (eds.). 1998. Rehabilitation of Rivers: Principles and Implementation. John Wiley and Sons, Chichester, U.K.

Dietz, M., and J. Clausen. 2006. Saturation to Improve Pollutant Retention in a Rain Garden. Environmental Science and Technology 40(4):1335-1340.

Dinicola, R.S. 1989. Characterization and Simulation of Rainfall-Runoff Relations for Headwater Basins in Western King and Snohomish Counties, Washington State. U.S. Geological Survey Water-Resources Investigation Report 89-4052.U.S. Geological Survey, Tacoma, WA.

Elsner, M.M., L. Cuo, N. Voisin, J.S. Deems, A.F. Hamlet, J.A. Vano, K.E.B. Mickelson, S.Y. Lee, and D.P. Lettenmaier. 2009. Implications of 21st Century Climate Change for the Hydrology of Washington State. In The Washington Climate Change Impacts Assessment, Elsner, M.M., J. Littell, and L.W. Binder (eds.). Center for Science in the Earth System, Joint Institute for Study of the Atmosphere and Oceans, University of Washington, Seattle, WA, pp. 69-106.

EnviroVision Corporation, Herrera Environmental Consultants, Inc., and Washington Department of Ecology (WDOE). 2008. Phase 2: Improved Estimates of Toxic Chemical Loadings to Puget Sound from Surface Runoff and Roadways. WDOE, Olympia, Washington.

Ermilio, J., and R. Traver. 2006. Hydrologic and Pollutant Removal Performance of a Bio-infiltration BMP. EWRI 2006, National Symposium.

Federal Highway Administration (FHWA). 2007. Design for Fish Passage at Roadway-Stream Crossings: Synthesis Report, U.S. Department of Transportation Publication No. FHWA-HIF-07-033. Federal Highway Administration, McLean, VA.

Federal Interagency Stream Restoration Working Group. 1998. Stream Corridor Restoration: Principles, Processes, and Practices. Federal Interagency Stream Restoration Working Group, Washington, DC.

Field, R., J.P. Heaney, and R. Pitt. 2000. Innovative Urban Wet-Weather Flow Management Systems. Technomic Publishing Company, Inc., Lancaster, PA.

Fore, L.S., J.R. Karr, and R.W. Wisseman. 1996. Assessing Invertebrate Responses to Human Activities: Evaluating Alternative Approaches. Journal of the North American Benthological Society 15(2):212-231.

Frei, C.M. 2006. Design of Fish Passage at Bridges and Culverts, Hydraulic Engineering Circular – 26. M.S.C.E. thesis, Washington State University, Pullman, WA

Frissell, C.A. 1997. Ecological Principles. In Williams, J.E., C.A. Wood, and M.P. Dombeck (eds.). 1997. Watershed Restoration: Principles and Practices. American Fisheries Society, Bethesda, MD, pp. 96-115.

Furniss, M.J., T.D. Roelofs, and C.S. Yee. 1991. Road Construction and Maintenance. In W.R. Meehan (ed.), Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats, American Fisheries Society, Special Publication 19, Bethesda, MD, pp. 297–324.

Garie, H.L. and A. McIntosh. 1986. Distribution of Benthic Macroinvertebrates in Streams Exposed to Urban Runoff. Water Resources Bulletin 22(3):447-455.

Geosyntec Consultants. 2008. Post-Construction BMP Technical Guidance Manual. City of Santa Barbara, CA.

Granger, T., T. Hruby, A. McMillan, D. Peters, J. Rubey, D. Sheldon, S. Stanley, and E. Stockdale. 2005. Wetlands in Washington State, Volume 2: Guidance for Protecting and Managing Wetlands. Washington Department of Ecology, Olympia, WA.

Hart Crowser, Inc. 2007. Phase 1: Initial Estimate of Toxic Chemical Loadings to Puget Sound. Washington State Department of Ecology, United States Environmental Protection Agency, and Puget Sound Partnership, Olympia, WA.

Heathwaite, L. A. Sharpley, and W. Gburek. 2000. A Conceptual Approach for Integrating Phosphorus and Nitrogen Management at Watershed Scales. Journal of Environmental Quality 29(1):158-166.

Helsel, D. R. and R.M. Hirsch. 1991. Statistical Methods in Water Resources: Techniques of Water-Resources Investigations, Chapter A3 of Book 4. U.S. Geological Survey, Reston, VA.

Hinman, C. 2005. Low Impact Development: Technical Guidance Manual for Puget Sound. Puget Sound Action Team, Olympia, WA.

Hinman, C. 2007. Rain Garden Handbook for Western Washington Homeowners: Designing Your Landscape to Protect Our Streams, Lakes, Bays, and Wetlands. Washington State University Extension Pierce County, Tacoma WA.

Horner, R.R., D.B. Booth, A. Azous, and C.W. May. 1997. Watershed Determinants of Ecosystem Functioning. In L.A. Roesner (ed.), Effects of Watershed Development and

Management on Aquatic Ecosystems, American Society of Civil Engineers, New York, NY, pp. 251-274.

Horner, R.R., A.L. Azous, K.O. Richter, S.S. Cooke, L.E. Reinelt, and K.Ewing. 2001. Wetlands and Stormwater Management Guidelines. In Azous, A.L. and R.R. Horner (eds.), Wetlands and Urbanization: Implications for the Future, Lewis Publishers, Boca Raton, FL, pp. 299-323.

Horner, R.R., C.W. May, and E.H. Livingston. 2003. Linkages Between Watershed and Stream Ecosystem Conditions in Three Regions of the United States. Report to U.S. Environmental Protection Agency by Watershed Management Institute, Inc., Crawfordville, FL.

Horner, R., C. May, E. Livingston, D. Blaha, M. Scoggins, J. Tims, and J. Maxted. 2002. Structural and Non-structural Best Management Practices (BMPs) for Protecting Streams. In B.K. Urbonas (ed.), Linking Stormwater BMP Designs and Performance to Receiving Water Impact Mitigation, American Society of Civil Engineers, New York, pp. 60-77.

Horner, R.R. and T. Osborne. 2005. Removal of Fecal Coliforms From Stormwater Runoff: A Literature Review. Report to City of Blaine, WA.

Horner, R.R. and C. Chapman. 2007. NW 110TH Street Natural Drainage System Performance Monitoring, With Summary of Viewlands and 2nd Avenue NW SEA Streets Monitoring. Report to Seattle Public Utilities by Department of Civil and Environmental Engineering, University of Washington, Seattle, WA.

Chapman, C. and R.R. Horner. 2010. Performance Assessment of a Street-Drainage Bioretention System. Water Environment Research 82(2): 109-119.

Hughes, F. M., R..A. Colston, and J.O. Mountford. 2005. Restoring Riparian Ecosystems: The challenge of Accommodating Variability and Designing restoration Trajectories. Ecology and Society 10(1):12.

Hunt, W.F., J.T. Smith, S.J. Jadlocki, J.M. Hathaway, and P.R. Eubanks. 2008. Pollutant Removal and Peak Flow Mitigation by a Bioretention Cell in Urban Charlotte, NC. Journal ofEnvironmental Engineering 134(5):403-408.

Hynes, H.B.N. 1970. Ecology of Running Waters. University of Toronto Press, Toronto, Ontario, Canada.

Hynes, H.B.N. 1974. The Biology of Polluted Waters. University of Toronto Press, Toronto, Ontario, Canada.

Jacobsen, P.S., K.K. Jones, and K.J. Anlauf. 2007. 2002-2005 Effectiveness Monitoring for the Western Oregon Stream Restoration Program. Monitoring Program Report Number OPSW-ODFW-2007-6, Oregon Department of Fish and Wildlife, Salem, OR.

James, C.S. 2007. Evaluation of Stream Restoration Projects in Washington State. M.S. thesis, College of Forest Resources, University of Washington, Seattle, WA.

Jantrania, A.R. and M.A. Gross. 2006. Advanced Onsite Wastewater Systems Technologies. CRC Taylor & Francis Group, Boca Raton, FL.

Johnson, P.A., D.L. Mock, E.J. Teachout, and A. McMillan. 2000. Washington StateWetland Mitigation Evaluation Study, Phase 1: Compliance. Washington StateDepartment of Ecology, Olympia, WA.

Johnson P., D.L. Mock, A. McMillan, L. Driscoll, and T. Hruby. 2002. Washington State Wetland Mitigation Evaluation Study, Phase 2: Evaluating Success. Washington Department of Ecology, Olympia, WA.

Karr, J.R., L.A. Toth, and D.R. Dudley. 1985. Fish communities of Midwestern Rivers: A History of Degradation. BioScience35:90–95.

Klein, R.D. 1979. Urbanization and Stream Quality Impairment. Water Resources Bulletin 15:948-963.

Konrad, C.P., and S.J. Burges. 2001. Hydrologic Mitigation Using On-site Residential Stormwater Detention. Journal of Water Resources Planning and Management127:99-107.

Larimore, R.W., and P.W. Smith. 1963. The Fishes of Champaign County, Illinois, as Affected by 60 years of Stream Changes. Bulletin of the Illinois Natural History Survey 28:299–382.

Limburg, K.E. and R.E. Schmidt. 1990. Patterns of Fsh Spawning in Hudson River Tributaries: Response to an Urban Gradient. Ecology 71(4):1238-1245.

Lloyd, S., T. Wong and C. Chesterfield. 2002. Water Sensitive Urban Design: A Stormwater Management Perspective. Cooperative Research Centre for Catchment, Monash University, Victoria, Australia.

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian Forests as Nutrient Filters in Agricultural Watersheds. BioScience 34(6):374-377.

Mantua, N., I. Tohver, and A. Hamlet. 2009. Impacts of Climate Change on Key Aspects of Freshwater Salmon Habitat in Washington State. InThe Washington Climate Change Impacts Assessment, Elsner, M.M., J. Littell, and L.W. Binder (eds.). Center for Science in the Earth System, Joint Institute for Study of the Atmosphere and Oceans, University of Washington, Seattle, WA, pp. 217-253.

Mar, B.W., R.R. Horner, J.F. Ferguson, D.E. Spyrikakis, and E.B. Welch. 1982. Summary—Washington State Highway Runoff Water Quality Study, 1977-1982. Washington State Department of Transportation, Olympia, WA.

May, C.W. 1996. Assessment of Cumulative Effects of Urbanization on Small Streams in the Puget Sound Lowland Ecoregion: Implications for Salmonid Resource Management. Ph.D. dissertation, University of Washington, Seattle, WA, U.S.A.

May, C.W., E.B. Welch, R.R. Horner, J.R. Karr, and B.W. Mar. 1997b. Quality Indices for Urbanization Effects in Puget Sound Lowland Streams. Washington Department of Ecology, Olympia, WA.

McBride, M. and D. Booth. 2005. Urban Impacts on Physical Stream condition: Effects on Spatial Scale, Connectivity, and Longitudinal Trends. Journal of the American Water Resources Association 6: 565-580.

McMahon, T.E. and G.F. Hartman. 1989. Influence of Cover Complexity and Current Velocity on Winter Habitat Use by Juvenile Coho Salmon. Canadian Journal of Fisheries and Aquatic Sciences 46:1551-1557.

Meals, D.W., E.A. Cassell, D. Hughell, L. Wooda, W.E. Jokela, and R. Parsons. 2008a. Dynamic Spatially Explicit Mass-Balance Modeling for Targeted Watershed Phosphorus Management: I. Model Development. Agriculture, Ecosystems and Environment 127:189–200.

Meals, D.W., E.A. Cassell, D. Hughell, L. Wooda, W.E. Jokela, and R. Parsons. 2008b. Dynamic Spatially Explicit Mass-Balance Modeling for Targeted Watershed Phosphorus Management: II. Model Application. Agriculture, Ecosystems and Environment 127:223–233.

Metcalf and Eddy, Inc (revised by G. Tchobanoglous, F.L. Burton, and H.D. Stensel). 2003. Wastewater Engineering: Treatment and Reuse. McGraw-Hill, Boston, MA.

Miller, D.E. and P.B. Skidmore. 2003. Establishing a Standard of Practice for Natural Channel Design Using Design Criteria. In Montgomery, D.R., S. Bolton, D.B. Booth, and L. Wall (eds.), Restoration of Puget Sound Rivers, University of Washington Press, Seattle, WA, pp. 340-360.

Miller, S.W., P. Budy, and J.C. Schmidt. 2010. Quantifying Macroinvertebrate Responses to In-Stream Habitat Restoration: Applications of Meta-Analysis to River Restoration.Restoration Ecology 18(1):8-19.

Mitsch, W.J. and J.G. Gosselink. 2007. Wetlands, 4th Edition. John Wiley and Sons, Inc., Hoboken, NJ.

Mitsch, W.J., J.G. Gosselink, C.J. Anderson, and L. Zhang. 2009. Wetland Ecosystems. John Wiley and Sons, Inc., Hoboken, NJ.

Montgomery, D.R., S. Bolton, D.B. Booth, and L. Wall (eds.). 2003. Restoration of Puget Sound Rivers. University of Washington Press, Seattle, WA.

Montgomery, D.R., J.M. Buffington, R.D. Smith, K.S. Schmidt, and G. Pess. 1995. Pool Spacing in Forest Channels. Water Resources Research13:1097–1105.

Montgomery, D.R., G.E. Grant, and K. Sullivan. 1995. Watershed Analysis as a Framework for Implementing Ecosystem Management. Water Resources Bulletin 31(3):369-386.

Morley, S.A. and J.R. Karr. 2002. Assessing and Restoring the Health of Urban Streams in the Puget SoundBasin. Conservation Biology 16:1498-1509.

Mostaghimi, S., S.W. Park, R.A. Cooke, and S.Y. Wang. 1997. Assessment of Management Alternatives on a Small Agricultural Watershed. Water Research. 31(8):1867-1878.

Mostaghimi, S., K.M. Brannan, T.A. Dillaha, and A.C. Bruggeman. 2001. Best Management Practices for Nonpoint Source Pollution Control: Selection and Assessment. In W.F. Ritter and A. Shirmohammadi (eds.), Agricultural Nonpoint Source Pollution: Watershed Management and Hydrology, Lewis Publishers, Boca Raton, FL.

Naiman, R.J. (ed.). 1992. Watershed Management: Balancing Sustainability and Environmental Change. Springer-Verlag, New York, NY.

Naiman, R.J. and H. Décamp. 1997. The Ecology of Interfaces:Riparian Zones. Annual Review of Ecology and Systematics28:621–658.

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian Ecology andManagement in the Pacific Coastal Rain Forest. BioScience50(11):996-1011.

National Research Council (NRC). 1992. Restoration of Aquatic Ecosystems: Science, Technology, and Public Policy. National Academies Press, Washington, DC.

National Research Council (NRC). 1999. New Strategies for America's Watersheds. The National Academies Press, Washington, DC.

National Research Council (NRC). 2001. Compensating for Wetland Losses Under the Clean Water Act. The National Academies Press, Washington, DC.

National Research Council (NRC). 2009. Urban Stormwater Management in the United States. The National Academies Press, Washington, DC.

Natural Resources Conservation Service (NRCS). 2007a. Stream Restoration Design, National Engineering Handbook, Part 654. NRCS, U.S. Department of Agriculture, Washington, DC.

Natural Resources Conservation Service (NRCS). 2007b. National Handbook of Conservation Practices. NRCS, U.S. Department of Agriculture, Washington, DC.

Nelson, K. 1999. The influence of Sediment Supply and Large Woody Debris on PoolCharacteristics and Habitat Diversity. M. Sc. Thesis. Department of Civil Engineering,University of Washington, Seattle, WA.

Nelson , K.C., M.A. Palmer, J.E. Pizzuto, G.E. Moglen, P.L. Angermeier, R.H. Hilderbrand, M. Dettinger, and K. Hayhoe. 2009. Forecasting the Combined Effects of Urbanization and Climate Change on Stream Ecosystems: From Impacts to Management Options. Journal of Applied Ecology46:154–163

Nilsson, C., R. Jansson, B. Malmqvist, and R.J. Naiman. 2007. Restoring Riverine Landscapes: The challenge of Identifying Priorities, Reference States, and Techniques. Ecology and Society12(1):16.

Novotny, V. and G. Chesters. 1981. Handbook of Nonpoint Pollution: Sources and Management. Van Nostrand Reinhold Company, New York, NY.

Oregon State University, Geosyntec Consultants, University of Florida, and Low Impact Development Center, Inc. 2006. Evaluation of Best Management Practices for Highway Runoff Control. National Cooperative Highway Research Program Report 565, Transportation Research Board, Washington, DC.

Ormerod, S.J. 2009. Climate Change, River Conservation and the Adaptation Challenge. Aquatic Conservation: Marine and Freshwater Ecosystems 19:609–613.

Palmer, M.A., D.P. Lettenmaier, N.L.Poff, S.L. Postel, B.Richter, and R.Warner. 2009. Climate Change and River Ecosystems: Protection and Adaptation Options. Environmental Management 44:1053–1068.

Palmer, M.A., H.L. Menninger, and E. Bernhardt. 2010. River Restoration, Habitat Heterogeneity and Biodiversity: A failure of theory or Practice? Freshwater Biology 55 (Suppl. 1):205–222.

Pedersen, E.R. and M.A. Perkins. 1986. The use of Invertebrate Data for Evaluating Impacts of Urban Runoff. Hydrobiologia 139:13-22.

Pionke, H.P., W.J. Gburek, and A.N. Sharpley. 2000. Critical Source Area Controls on Water Quality in an Agricultural Watershed Located in the Chesapeake Basin. Ecological Engineering 14:325–335.

Prince George's County. 1999. Low-Impact Development Design Strategies: An Integrated Design Approach. Prince George's County, Largo, MD.

Puget Sound Partnership PSP). 2009. Using Results Chains to Develop Objectives and Performance Measures for the 2008 Action Agenda. Puget Sound Partnership, Olympia, WA.

Puget Sound Partnership (PSP). 2010. 2009 State of the Sound Report. PSP, Olympia, WA.

Puget Sound Action Team (PSAT). 2003. Natural Approaches to Stormwater Management:Low Impact Development in Puget Sound. Puget Sound Action Team, Olympia, WA.

Rao, N.S., Z.M. Easton, E.M. Schneiderman, M.S. Zion, D.R. Lee, and T.S. Steenhuis. 2009. Modeling Watershed-Scale Effectiveness of Agricultural Best Management Practices to Reduce Phosphorus Loading. Journal of Environmental Management 90:1385–1395.

Rashin, E. and C. Graber. 1992. Effectiveness of Washington's Riparian Management Zone Regulations for Protection of Stream Temperature. Washington Department of Ecology, Olympia, WA.

Rashin, E. and C. Graber. 1993. Effectiveness of Best Management Practices for Aerial Application of Forest Pesticides. Washington Department of Ecology, Olympia, WA.

Rashin, E., C. Clishe, A. Loch, and J. Bell. 1999. Effectiveness of Forest Road and Timber Harvest Best Management Practices with Respect to Sediment-Related Water Quality Impacts. Washington Department of Ecology, Olympia, WA.

Reimold, R.J. 1998. Watershed Management: Practice, Policies, and Coordination. McGraw-Hill, New York, NY.

Reinelt, L.E., R.R. Horner, and A. Azous. 1998. Impacts of Urbanization on Palustrine (Depressional Freshwater) Wetlands: Research and Management in the Puget Sound Region. Urban Ecosystems 2:219-236.

Reinelt, L.E., B.L. Taylor, and R.R. Horner. 2001. Morphology and Hydrology. In Azous, A.L. and R.R. Horner (eds.), Wetlands and Urbanization: Implications for the Future, Lewis Publishers, Boca Raton, FL, pp. 31-46.

Reinelt, L.E. and B.L. Taylor. 2001. Effects of Watershed Development on Hydrology. In Azous, A.L. and R.R. Horner (eds.), Wetlands and Urbanization: Implications for the Future, Lewis Publishers, Boca Raton, FL, pp. 221-236.

Richey, J.S. 1982. Effects of Urbanization on a Lowland Stream in Western Washington. Ph.D. dissertation, University of Washington, Seattle, WA, U.S.A.

Robison, E.G. 1999. Restoring Fish Passage at Road Crossings. In D. Shively and B. Streif (eds.), Practical Approaches to Restoring Watersheds, American Fisheries Society, Oregon Chapter, Sun River, OR, pp. 46–50.

Roni, P., T.J. Beechie, R.E. Bilby, F.E. Leonetti, M.M. Pollack, and G.R. Pess. 2002. A Review of Stream Restoration Techniques and a HierarchicalStrategy for Prioritizing Restoration in PacificNorthwest Watersheds. North American Journal of Fisheries Management22:1–20.

Rosenberg, E.A., P.W. Keys, D.B. Booth, D. Hartley, J. Burkey, A.C. Steinemann, and D.P. Lettenmaier. 2009. Precipitation Extremes and the Impacts of Climate Change on Stormwater Infrastructure in Washington State. InThe Washington Climate Change Impacts Assessment, Elsner, M.M., J. Littell, and L.W. Binder (eds.). Center for Science in the Earth System, Joint

Institute for Study of the Atmosphere and Oceans, University of Washington, Seattle, WA, pp. 311-343.

Rosgen, D. 1996. Applied River Morphology. Wildlife Hydrology, Pagosa Springs, CO. Rushton, B. 2002. Low Impact Parking Lot Design Infiltrates Stormwater. Florida Department of Environmental Protection, Tallahassee, FL.

Saldi-Caromile, K., K. Bates, P. Skidmore, J. Barenti, and D. Pineo. 2004. Stream Habitat Restoration Guidelines: Final Draft. Co-published by the Washington Departments of Fish and Wildlife and Ecology and the U.S. Fish and Wildlife Service, Olympia, WA. Schueler, T., D. Hirschman, M. Novotney, and J. Zielinski. 2007. Urban Subwatershed Restoration Manual No. 3: Urban Stormwater Retrofit Practices (Version 1.0), Center for Watershed Protection, Ellicott City, MD.

Scott, J.B., C.R. Steward, and Q.J. Stober. 1986. Effects of Urban Development on Fish Population Dynamics in Kelsey Creek, Washington. Transactions of the American Fisheries Society 115:555-567.

Sharkey, L.J. 2006. The Performance of Bioretention Areas in North Carolina: A Study of Water Quality, Water Quantity, and Soil Media. Thesis, North Carolina State University, Raleigh, NC.

Sharpley, A.N., R.W. McDowell, and P.J.A. Kleinman. 2001. Phosphorus Loss from Land to Water: Integrating Agricultural and Environmental Management. Plant and Soil 237:287–307.

Sharpley, A.N., J.L. Weld, D.B. Beegle, P.J.A. Kleinman, W.J. Gburek, P.A. Moore, Jr., and G. Mullins. 2003. Development of Phosphorus Indices for Nutrient Management Planning Strategies in the United States. Journal of Soil and Water Conservation 58(3):137-152.

Sheldon, D., T. Hruby, P. Johnson, K. Harper, A. McMillan, T. Granger, S. Stanley, and E. Stockdale. 2005. Wetlands in Washington State, Volume 1: A Synthesis of the Science. Washington Department of Ecology, Olympia, WA.

Smith, R.A., and W.F. Hunt. 2007. Pollutant Removal in Bioretention Cells With grass Cover. Proceedings of the World Environmental and Water Resources Congress.

Somerville, D.E. and B.A. Pruitt. 2004. Physical Stream Assessment: A Review of Selected Protocols for Use in the Clean Water Act Section 404 Program. U.S. Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Wetlands Division (Order No. 3W-0503-NATX). Washington, D.C.

Steedman, R.J. 1988. Modification and Assessment of an Index of Biotic Integrity to Quantify Stream Quality in Southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences45:492–501.

Steiner, G.R., J.T. Watson, and K.D. Choate. 1991. General Design, Construction, and Operation Guidelines: Constructed Wetlands Wastewater Treatment Systems for Small Users Including Individual Residences. Tennessee Valley Authority, Knoxville, TN.

Steiner, G.R. and D.W. Combs. 1993. Small Constructed Wetland Systems for Domestic Wastewater Treatment and Their Performance. In G.A. Moshiri (ed.), Constructed Wetlands for Water Quality Improvement, Lewis Publishers, Boca Raton, FL, pp. 491-498.

Stewart-Koster, B., S.E. Bunn, S.J .Mackay, N.L. Poff, R.J. Naiman, and P.S. Lake. 2010. The Use of Bayesian Networks to Guide Investments in Flow and Catchment Restoration for Impaired River Ecosystems. Freshwater Biology 55:243–260.

St. John, M.S. and R.R. Horner. 2007. Effect of Road Shoulder Treatments on Highway Runoff Water Quality. Washington State Department of Transportation, Olympia, WA.

Stouder, D.J., P.A. Bisson, and R.J. Naiman (eds.). 1997. Pacific Salmon and Their Ecosystems: Status and Future Options. Chapman and Hall, New York, NY. Strahler, A.N. 1952. Dynamic Basis of Geomorphology. Geological Society of America Bulletin 63:923-938.

Strecker, E.W., M.M. Quigley, B.R. Urbonas, J.E. Jones, and J.K. Clary. 2001. Determining Urban Storm Water BMP Effectiveness. Journal of Water Resources Planning and Management 127(3):144-149.

Struck, S.D., A.C. Rowney, and L.D. Pechacek. 2009. Innovative Approaches for Urban Watershed Wet-Weather Flow Management and Control: State-of-the-Technology: Interim Report. U.S. Environmental Protection Agency, National Risk Management Research Laboratory, Cincinnati, OH.

Taylor Associates, Inc. 2008. Boatyard Stormwater Treatment Technology Study. Report to Northwest Marine Trade Association, Puget Soundkeeper Alliance, and Washington Department of Ecology by Taylor Associates, Inc., Seattle, WA.

Terrene Institute. 1996. A Watershed Approach to Urban Runoff: Handbook for Decisionmakers. Terrene Institute, Washington, D.C.

Tramer, E.J., and P M. Rogers. 1973. Diversity and Longitudinal Zonation in Fish Populations of Two Streams Entering a Metropolitan Area. American Midland Naturalist90:366–374.

Trautman, M.B., and D.K. Gartman. 1974. Re-evaluation of the Effects of Man-Made Modifications on Gordon Creek between 1887 and 1973 and Especially as Regards its Fish Fauna. Ohio Journal of Science74:162–173.

U.S. Environmental Protection Agency (USEPA). 1983. Results of the Nationwide Urban Runoff Program: Volume 1 - Final Report.USEPA, Water Planning Division, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2000. Manual, Constructed Wetlands Treatment of Municipal Wastewaters. USEPA, National Risk Management Research Laboratory, Cincinnati, OH.

U.S. Environmental Protection Agency (USEPA). 2002. Onsite Wastewater Treatment System Manual. USEPA, Office of Water, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2003a. Watershed-Based NPDES Permitting Policy Statement. In Watershed-Based National Pollutant Discharge Elimination System (NPDES) Permitting Implementation Guidance. USEPA, Washington, DC.

U.S Environmental Protection Agency (USEPA). 2003b. Watershed-Based National Pollutant Discharge Elimination System (NPDES) Permitting Implementation Guidance. USEPA, Washington, DC.

U.S Environmental Protection Agency (USEPA). 2003c. National Management Measures for the Control of Nonpoint Pollution from Agriculture. USEPA, Office of Water, Washington, DC.

U.S Environmental Protection Agency (USEPA). 2005. National Management Measures to Control Nonpoint Source Pollution from Forestry. USEPA, Nonpoint Source Control Branch, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2007a. Watershed-Based NPDES Permitting Technical Guidance (draft). USEPA, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2007b. Reducing Stormwater Costs through Low ImpactDevelopment (LID) Strategies and Practices. USEPA, Nonpoint Source Control Branch, Washington, DC.

U.S. Environmental Protection Agency (USEPA). 2007c. Advanced Wastewater Treatment to Achieve Low Concentration of Phosphorus. USEPA, Region 10, Seattle, WA.

U.S. Environmental Protection Agency (USEPA). 2008. Managing Wet Weather with Green Infrastructure Action Strategy. USEPA, Washington, DC.

Van Seters, T., D. Smith and G. MacMillan. 2006. Performance Evaluation of Permeable Pavement and a Bioretentions Swale. Proceedings 8th International Conference on Concrete Block Paving, San Fransisco, CA.

Washington Department of Ecology (WDOE). 1991. Summary of Criteria and Guidelines for Contaminated Freshwater Sediments. WDOE, Olympia, WA.

Washington Department of Ecology (WDOE). 2005. Stormwater Management Manual for Western Washington. WDOE, Olympia, WA.

Washington Department of Ecology (WDOE). 2006a. Wetland Mitigation in Washington State, Part 1: Agency Policies and Guidance. WDOE, Olympia, WA.

Washington Department of Ecology (WDOE). 2006b. Wetland Mitigation in Washington State, Part 2: Developing Mitigation Plans. WDOE, Olympia, WA.

Washington Department of Ecology (WDOE). 2008. Criteria for Sewage Works Design. WDOE, Olympia, WA.

Washington Department of Fish and Wildlife (WDFW). 2003a. Integrated Streambank Protection Guidelines. Washington State Aquatic Habitat Guidelines Program, in association with WDFW, Washington State Department of Transportation, and Washington Department of Ecology, Olympia, WA.

Washington Department of Fish and Wildlife (WDFW). 2003b. Design of Road Culverts for Fish Passage. WDFW, Olympia, WA.

Washington State Department of Health (WSDOH). 2005. Nitrogen Reducing Technologies for Onsite Wastewater Treatment Systems. Washington State Department of Health, Olympia, WA. Water Environment Federation (WEF). 2006. Membrane Systems for Wastewater Treatment. WEF Press, McGraw-Hill, New York, NY.

Weaver, L.A. and G.C. Garman. 1994. Urbanization of a Watershed and Historical Changes in a Stream Fish Assemblage. Transactions of the American Fisheries Society 123:162-172.

Weinstein, N., C. Glass, J. Heaney, W. Huber, P. Jones, C. Kloss, M. Quigley, K. Stephens, and E. Strecker. 2006. Decentralized Storm-water Controls for Urban Retrofit and Combined Sewer Overflow Reduction. Water Environment Research Foundation, Alexandria, VA.

Weinstein, N, J. Cotting, D. Nees, S. Downing, J. Lee, B. Tauber, A. English, C. Kloss, C. Glass, W.C. Huber, and T. Morgan. 2009. Decentralized Stormwater Controls for Urban Retrofit and Combined Sewer Overflow Reduction, Phase 2. Water Environment Research Foundation, Alexandria, VA.

Welch, E.B. and J.M. Jacoby. 2004. Pollutant Effects in Freshwater: Applied Limnology, 3rd Ed. Spon Press, London, U.K.

Williams, J.E., C.A. Wood, and M.P. Dombeck (eds.). 1997. Watershed Restoration: Principles and Practices. American Fisheries Society, Bethesda, MD. Wissman, R.C. and P.A. Bisson (eds.). 2003. Strategies for Restoring River Ecosystems: Sources of Variability and Uncertainty in Natural and Managed Systems.American Fisheries Society, Bethesda, MD

Marine and Estuarine Protection and Restoration Strategies

This section focuses on the scientific basis for a suite of marine, nearshore, and estuarine protection and restoration strategies. The strategies addressed come from a number of sources including the Puget Sound Partnership Action Agenda (PSP 2009), the Puget Sound Nearshore Ecosystem Restoration Project (e.g., Clancy et al. 2009), and other existing state, federal, and tribal programs. The strategic topics addressed in this section are generally grouped by 1) Puget Sound water quality and 2) physical habitat protection, restoration, and management processes. Each strategy is evaluated on its scientifically demonstrated effectiveness, level of certainty, and/or gaps in science-based knowledge, based on thorough review of the literature. The strategies include ways to comprehensively manage/integrate all natural processes and human activities that involve salt and freshwater (infiltration, recharge, surface runoff, collection, storage, diversion, transport and use), effluent, wastewater treatment, point and non-point pollution, spills, and discharge at appropriate temporal and spatial scales, many of which are covered in Section 3. The ultimate goal is to replicate and maintain as much as possible the functional characteristics (quality, quantity, rates, connectivity) of the natural system at all appropriate scales, times, and places.

1. Background

There are two primary sources of water flowing into the Puget Sound: tidally driven marine water mixing in from the Pacific Ocean and freshwaters entering from rivers, streams, surface flow, and groundwater discharge. Rivers and streams at times deliver excessive nutrients, sediments, toxic contaminants, pathogens, and freshwater to Puget Sound. Watershed protection and restoration strategies are intended to result in improved water quality of freshwater rivers and streams entering Puget Sound estuaries and marine waters. These topics are covered in Section 3 and will not be repeated here. Therefore, water quality topics addressed in this section apply only to surface runoff, groundwater discharges, and effluents that drain directly into estuaries or marine waters along Puget Sound shorelines and to other water quality issues in Puget Sound proper, many of which are also covered in Section 3.

Nutrient Loading

WDOE High Nitrogen Study (WDOE 2008) summarizes the nitrogen input pattern for southern Puget Sound (see Chapter 2A and Section 3 of this Chapter).

Contaminant Loading

Some areas of Puget Sound have excessive contaminants in the water and sediments. The array of contaminants in Puget Sound includes heavy metals, PAHs, PDBEs, PCBs, dioxins, phthalates, pharmaceuticals, cosmetics, and other personal care products (Hart Crowser, Inc, et al. 2007). The primary sources of contaminants in Puget Sound are from surface runoff, atmospheric deposition, industrial and municipal waste waters, combined sewer overflows (CSOs), and direct spills (Hart Crowser, Inc., et al. 2007; see Chapters 2A and 3 of the PSSU).

Because of the challenges associated with reducing sediment contaminant loads in deep water, we focus on reducing the amount of contaminants delivered to Puget Sound. General strategies include reducing contaminants in treatment plant effluents, preventing contaminants spills, and cleaning up known sources of contamination.

One strategy that could help to reduce contaminant loading is to use a toxic loading inventory to guide loadings reduction strategies (e.g., Paulson et al. 1989, Hart Crowser, Inc. et al. 2007, EnviroScience Corp. et al. 2008). Many restoration strategies for reducing contaminant inflows are similar to those for reducing nutrient loads (e.g., wastewater treatment, reducing storm water, on-site treatment). Toxic spill prevention and cleanup are additional strategies that pertain to contaminants.

Improving Wastewater Treatment Plants that Drain Directly into Estuaries or Puget Sound

Wastewater treatment has a long history, based initially on common sense. The first treatment systems consisted primarily of flushing waste away from human population centers with water flow, often downstream to larger rivers and ultimately marine waters. We now know that when effluent is highly concentrated, not dispersed by tidal currents, and/or contains high concentrations of deleterious constituents, problems arise in the human and natural environment (e.g., Malins 1984, McCain et al. 1988). Effects of wastewaters on Puget Sound have been discussed in previous Puget Sound science update sections and the majority of wastewater treatment restoration strategies have been discussed in Section 3. In this section we discuss wastewater protection and restoration strategies that are either not covered in Section 3 or are particularly relevant to Puget Sound proper. They are: 1) Combined sewer overflows 2) Programs to address heavy nutrient loading of South Puget Sound and Hood Canal. 3) Reducing toxic loads in Puget Sound, 4) Preventing and reducing the effects of wastewater constituents that are not fully treated such as pharmaceuticals, cosmetics, cleaning products, industrial materials, etc. and 5) Water reuse as a restoration strategy.

Municipal and industrial wastewater treatment plants discharge effluent directly into Puget Sound in a number of locations, most notably West Point in Seattle, Snohomish Estuary and Port Gardner in Everett, and Budd Inlet in Olympia and others. These facilities receive much of the Puget Sound area municipal waste waters as well as permitted industrial effluent. Industrial facilities typically have systems customized to their waste products and sometimes discharge to municipal systems following pre-treatment. The treatment systems remove the majority of solids, biodegradable organics, and pathogens from the wastewater but they do not eliminate the high nitrogen loads from the effluent, nor do they fully remove many other toxics constituents such as heavy metals, pharmaceuticals, and PAHs, among many others (for details, see http://www.ecy.wa.gov/programs/wq/permits/northwest_permits.html). See Section 3 for a complete review of municipal wastewater restoration strategies.

Expanding and updating wastewater treatment facilities

The Puget Sound action agenda emphasizes need for expanding and updating wastewater treatment facilities (PSP 2009). The benefits from this restoration strategy have been described in

Section 3. The essence of this restoration strategy is to implement wastewater technology that maximizes the concept of secondary and tertiary treatment, including removal of all constituents that occurred effluent greater than background levels. The action agenda has prioritized expansion and updating of wastewater treatment facility at the highest level (PSP 2009).

Advanced Wastewater Treatment Nutrient Reduction

Reduction of anthropogenic nitrogen loads in Puget Sound will depend on a combination of treatment approaches that include advanced wastewater treatment in plants that discharge into both rivers and Puget Sound proper. The details of advanced wastewater treatment are addressed in Section 3.

Contaminant Reduction

Heavy metals and other contaminants (e.g., PAHs, PDBEs, PCBs, dioxins, phthalates) are known to be accumulating in Puget Sound (Hart Crowser, Inc., et al. 2007, EnviroScience Corp. et al. 2008). Wastewater treatment only partially removes contaminants, depending on the process used and the target contaminant.

Some treatment processes remove heavy metals in varying degrees. The advanced treatment process of reverse osmosis system is effective in removing some metals, such as arsenic, cadmium, chromium, and mercury to safe levels (WEF 2006). New technologies hold promise for future improvements in heavy metal removal from effluents (e.g., Sayari et al. 2005), but the applications of these improvements in Puget Sound treatment plants is unclear[1].

The relative treatment efficiencies for pharmaceuticals and personal care products (PPCPs) at five municipal wastewater treatment plants (WWTPs) in the Pacific Northwest were evaluated by Lubliner et al. (2010) and found to be mixed. Wastewater influent, secondary effluent, tertiary effluent, and biosolids were sampled. Four of the five WWTPs discharge within the Puget Sound watershed. Two of the plants provide secondary treatment, and three employ advanced (tertiary) treatment for nitrogen and phosphorus removal. Two of the plants produce tertiary-treated reclaimed water. Target analytes included 172 organic compounds (PPCPs, hormones, steroids, semi-volatile organics). Newly approved EPA methods were used to measure PPCPs, hormones, and steroids at low concentrations. Removal efficiencies were evaluated for each analyte at the five WWTPs. Secondary treatment alone achieved high removals for hormones and steroids. Approximately 21% of the 172 analytes were reduced to below reporting limits (i.e., 79% were not) by conventional secondary treatment, whereas 53% were reduced to below reporting limits by at least one advanced nutrient-removal technology. Roughly 20% of the 172 analytes (mainly polycyclic aromatic hydrocarbons) were found only in the biosolids and not the wastewater samples, so some analytes were clearly concentrating in the biosolids. Three PPCPs (carbamazepine, fluoxetine, and thiabendazole) were relatively untreated by the surveyed WWTP technologies. These three PPCPs may serve well as human-influence tracer compounds in the environment. Overall, the screening study indicates that (1) there are differences in PPCP removal between the WWTP processes and (2) advanced nutrient reduction and tertiary filtration may provide additional PPCP removal (Lubliner et al. 2010). A summary of the Department of

Ecology program for control of toxic pollutants in Puget Sound is found at http://www.ecy.wa.gov/programs/wq/pstoxics/index.html.

Combined Sewer Overflows

Combined sewer overflows (CSOs) are a concern because untreated wastewater and stormwater may be discharged to Puget Sound during large storms posing risks to public health and the environment. Details on strategies for reducing CSOs can be found in Section 3.

Linking outlet quantities with nutrient and contaminant dispersal

One strategy for reducing the effects of wastewater effluents on receiving waters has been to relocate discharge pipes into areas that are more conducive to dispersal. When discharges enter shallow, closed embayments with low flushing rates there is a tendency for contaminants and nutrients to build up. There is substantial scientific and technical basis for the strategy of locating outfalls at locations and depths that maximize diffusion and therefore minimize physical and biological effects of high concentrations of nutrients and contaminants.

A further subcomponent of outfall relocation is to use the various permutations of diffusers and/or depth as techniques to increase dispersal of effluent. This is done by expanding mixing zones, and hence enabling increased total toxic pollutant load, through engineered changes to effluent outfalls (e.g., lengthening of discharge outfalls by adding diffuser ports). Outfalls have advanced from simple open-ended pipes not far from shore to long outfalls with large multiple-port diffusers discharging in deep water. An example of this in Puget Sound is the extreme dimensions of King County's Brightwater project outfall: extending one mile offshore, at 600 feet deep, off of Point Wells in Puget Sound (see http://www.kingcounty.gov/environment/wtd/Construction/North/Brightwater.aspx). However, in other locations, outfalls have been constructed with much greater dimensions such as in Boston where one outfall is 9.4 miles long, 24.2 feet in diameter, including a 6,600 foot long diffuser section with 55 vertical risers, each with 8 discharges ports (NRC 1993).

The design of diffusion ports also has an important effect on the potential concentration of contaminants, especially in the sediments. Diffusers that lie on or near the bottom sediments will tend to concentrate certain contaminants more readily than diffusers that have vertical risers. The performance of a variety of diffuser configurations can be evaluated via a modeling environment (e.g., Roberts et al. 1989).

The optimal placement and configurations of effluent outfalls can be determined in concert with the interplay of ambient current patterns using models (e.g., Baumgartner et al. 1994, Frick 2003). In Puget Sound, such analyses could be conducted for both existing and proposed outfalls to determine the best locations and engineering design for either new outfalls or to retrofit existing outfalls.

Caution should be raised in terms of restoring water quality in Puget Sound through effluent relocation and redesign alone since this would likely result in simply expanding contamination into new areas of Puget Sound bays and estuaries. Restoration will therefore depend on a combination of reducing toxic constituents, nutrient loads, and total volume of effluent, as well as appropriate strategic placement and design of outfalls. The reduction of wastewater loading to Puget Sound is currently part of the Action Agenda (C3), but there is no specific reference to relocating or redesigning outfalls commensurate with state-of-the-art outfall design (PSP 2009).

On-site Wastewater Treatment

This topic has been covered extensively in Section 3 but it is important to note that on-site wastewater treatment is a critical restoration strategy for Puget Sound proper, especially in certain areas where high nutrient loads are contributing to nitrification such as southern Puget Sound and Hood Canal[2].

Reclaimed water

An important emerging strategy relative to wastewater treatment that may be important for the health of Puget Sound and its watersheds is water reclamation1. The basic concept is to clean water sufficiently so that it can be used in municipal, agricultural, and industrial processes or infiltrated back into the natural system. One of the main benefits of reclaiming water is that ultimately less total water may be needed for human use, thereby freeing water that can remain in streams for fish and other aquatic life, as well as recreation.

Potential Effectiveness and Uncertainties in Wastewater Management

There has been extensive research on the effects of wastewaters on marine waters, and substantial review of the effectiveness of wastewater treatment on freshwaters (see Section 3), but less research has been conducted on the effectiveness of wastewater treatment in marine waters. From a marine ecosystem health perspective, the ultimate goal is to reduce nutrients and contaminants to safe levels. It may be technically possible to eliminate harmful constituents from wastewater; the few exceptions include processes for reducing some heavy metals and some pharmaceuticals and personal care products -- more research is clearly needed in this area[3]. Nevertheless, the key question is whether the return on the investment will be effective. The certainty that these activities will be technically effective is very high. The uncertainty comes from policy decisions, availability of funding, and fully functioning monitoring program that can determine if recovery goals are being met.

Programs to Reduce Stormwater Run-off Directly into Estuaries and Puget Sound

As discussed in Section 3, stormwater can deliver heavy loads of nutrients, pathogens, toxic contaminants, and sediment to Puget Sound bays and estuaries, adding significantly to the total loads from all sources. Mercury, PCBs, flame retardants, and other persistent chemicals are found throughout Puget Sound where they they can bioaccumulate and transfer through the food web (see Chapter 2A of the Puget Sound Science Update and Section 3 of Chapter 4).

Accidental or Long-term Contaminant Spills

Programs and regulations that prevent shoreline- and boat-based accidental contaminants spills

The most obvious strategy for protecting marine waters against contamination from accidental spills of toxic substances is through spill prevention. There are numerous specific spill prevention activities. Many have focused on preventing bulk oil spills but others pertain to hazardous chemicals in transit, industrial use, wastewater from system shutdown or storm overflow and fuel spills from marine accidents. Particularly insidious are small, gradual, chronic releases of contaminants from diverse sources.

In Puget Sound, the major spill prevention programs are coordinated by the Department of Ecology[4]. Critical aspects of the program are preparedness, pre-booming, a system for advanced notification of oil transfer, containment requirements, spill drills, and the Puget Sound Safety Plan. Each one of these components plays a role in prevention, but some of them, like preparedness and response, also comprise the system for responding to spills when they happen (and are therefore addressed below).

The Puget Sound Harbor Safety Plan includes guidance to avoid a variety of navigational risks and hazards including aids to navigation, advanced notice of arrival, automatic identification system, required charts, emergency response communications, fishing net conflicts resolution, naval vessel protection zones, avoidance of marine sanctuaries, pilotage, and small vessel and marine that management (Puget Sound Harbor Safety Committee 2008). The Plan also includes Standards of Care that, taken together, all lead to safer operational conditions that can prevent the likelihood of marine contaminant spills. The Plan addresses procedures for anchoring, operations near bridges, bunkering, equipment failures, heavy weather, hot work, lightering, propulsion loss prevention, restricted visibility, tanker escort operations, towing vessel operations, and under-keel clearance (Puget Sound Harbor Safety Committee 2008).

Enforcement of spill prevention regulations is an integral part of successful spill prevention. Another quasi-enforcement concept that probably lends itself to spill prevention is public recognition of corporations as good citizens (e.g., Konar and Cohen 1997). The Department of Ecology Spill Prevention Program also includes guidance to limit discharges of unwanted materials from cruise ships and guidelines for ballast water management to protect from invasive species.

Clean-Up of Contaminants

Cleanup ranges from major EPA Superfund sites to clean up of minor spills. In some cases hazardous materials have been on-site for decades and have been or will be cleaned up and sites remediated while contemporary spills are usually cleaned up immediately or soon after spills.

Effectiveness of cleanup generally depends on 1) the amount of product released, 2) the contaminants were released, 3) chemical composition of the hazardous materials, 4) the specific technology of cleanup for each contaminant, 5) effectiveness of the cleanup technology, 6) the area or extent of the spill, and 7) the dispersal modes and rates (e.g., Etkin 2009).

Depending on the contaminants toxins involved and who caused the spill and where it occurred, contemporary cleanup response is managed by a coordinated effort of federal, state, tribal, and/or local agencies and the private sector. Basic policies for this coordination are set forth in the National Incident Management System (NIMS) of FEMA[5]. Spills anywhere in the country can be reported through the National Response Center[6]. In Washington, the response system is stepped down to the Department of Ecology's Incident Command System (ICS)[7].

Spill preparedness involves a continuous cycle of activities, capturing lessons learned and then incorporating them back into plans, policies and procedures. The cycle is necessary to promote coordination among a combination of the variety of entities involved, all using the ICS. Spill preparedness includes the following topics[8]:

- *Contingency Plans*. The Washington Administration Code (WAC 173-182) requires certain oil handling facilities, pipelines, and vessels to have a state-approved oil spill Contingency Plan that ensures their ability to respond to major oil spills.
- *Oil spill drills* enable response personnel to become knowledgeable and proficient in the strengths and weaknesses of plans, equipment and procedures. Oil spill drills are scheduled in advance on the area drill calendar. Ecology tracks drill progress over a three year cycle and has prepared a drill manual to assist in meeting the requirements of the drill program.
- *Primary Response Contractors* (PRCs) are private companies or cooperatives that are in partnership with Plan holders to act as required response support teams. To be cited by a plan holder, the contractor must apply and be approved by the Department of Ecology. The PRCs have equipment and crews that are trained and equipped to mitigate leaks and spills when they occur. The need to respond as soon as possible, with trained operators and systems of equipment that are enhanced for maximum effectiveness, is critical to increase the opportunity for on-water recovery and reduce shoreline oiling.
- *Geographic Response Plans* (GRPs) are site-specific response plans for oil spills to water. They include response strategies tailored to a specific beach, shore, or waterway and meant to minimize impact on sensitive areas threatened by the spill. Each GRP has two priorities, which are to: 1) Identify sensitive natural, cultural or significant economic resources; and 2) Describe and prioritize response strategies.
- *Incident Command System* (ICS) is a standardized on-scene emergency management system specifically designed to allow its user(s) to adopt an integrated organizational structure equal to the complexity and demands of single or multiple incidents, without being hindered by jurisdictional boundaries.
- *PRC equipment maps* depict the location of oil spill response equipment that is owned and operated by the state’s approved response contractors or oil spill contingency planners (industry). The maps include the locations of booms and skimmers and the capacity of each.
- *Trajectory Analysis Planner* (TAP) is a computer-based tool that investigates the probabilities that spilled oil will move and spread in particular ways within a particular area. TAP does this by assessing hundreds of site-specific spill trajectories. The Puget Sound TAP Technical Document describes the TAP methodology and trajectory modeling behind TAP, as well as the accuracy and limitations of TAP.

Once a spill occurs, the progress of response and cleanup is tracked on the Department of Ecology's Spills website (http://www.ecy.wa.gov/programs/spills/incidents/main.html). The scientific basis of contaminant cleanup is extensive, especially for spilled petroleum products; there is less extensive scientific guidance for cleaning up other contaminants. Several key books on the extensive science of oil's effects on the environment, spill prevention and preparedness, and the techniques of cleanup are Lane (1995), Cormack (1999), and Ornitz and Champ (2002).

Clean-up of historic marine/estuarine industrial toxic waste sites

There are a wide variety of toxic cleanup sites that affect Puget Sound. Some include contaminants that were released years ago and the others are from more recent spills or chronic pollution problems. Because of the wide variation in on-site conditions and the contaminants that require cleanup, the cleanup process at each location is engineered case-by-case. In some locations, a bay-wide approach is taken to clean up, especially for toxins that were delivered from multiple sources but deposited in the sediments of the same area. Under the Puget Sound Initiative (PSI), the Department of Ecology has prioritized certain bays and organize cleanup in some locations by the bay-wide approach. The PSI includes all toxic waste sites within 1/2 mile of Puget Sound.

In the Puget Sound, the US EPA has the lead on federally-listed hazardous waste sites and Ecology has the lead on state clean up sites. There are many steps between discovery of a toxic site requiring cleanup and the final cleanup including initial investigation, site hazardous assessment, site ranking and listing, emergency actions if necessary, feasibility study, cleanup action plan, engineering design, cleanup construction, cleanup operation and maintenance, environmental covenants, periodic reviews, and finally, removal from hazardous sites list. The physical core of this restoration strategy is the construction phase (i.e., actions taken at a site to eliminate, render less toxic, stabilize, contain, immobilize, isolate, treat, destroy, or remove a hazardous substance). These generally include construction activities such as removal of contaminated soils or sediment for off-site treatment or disposal; pumping and treating of contaminated ground water; sealing off contaminated soils or sediment beneath a cap or barrier; the addition of chemicals or enhancement of the growth of microorganisms that break down contaminants in place[9]; etc. Specifics of each toxin remediation project can be found on the Department of Ecology's website[10].

Other critical considerations in toxic cleanup as a restoration strategy are the legal process and financial responsibility. The process for identifying responsible parties and coming to agreement on cleanup costs can be long and arduous. In addition to federal laws, basic legal vehicles for cleanup enforcement and associated Washington Administrative Code references are cited on the Department of Ecology website[11].

Creating new habitat as part of hazardous waste cleanup is a restoration strategy that can add environmental and social benefit during recovery. One local example is at the Commencement Bay Asarco Superfund site where NOAA fisheries worked to include habitat features with the site remediation process. EPA supports the related Brownfields program that is designed to create new habitats and mediated sites pre-building development especially for community benefits[12].

The scientific basis for toxic waste clean-up is extensive, but somewhat lacking in many technical areas. Several key books about scientific cleanup techniques and technologies are NRC (1995), Boulding (1996), Sellers (1998), and Lehr et al. (2001). NRC (1995) primarily evaluated current management practices and technologies for cleanup. They also cite, among the many technical challenges to be overcome in managing contaminated sediments, are an inadequate understanding of the natural processes governing sediment dispersion, the bioavailability of contaminants, and technical difficulties involved in sediment characterization, removal, containment, and treatment. Sellers (1998) is a comprehensive guide for numerous hazardous waste site cleanup procedures. Lehr et al. (2001) cover many of the techniques for cleanup of environmental hazards in marine waters and adjacent shorelands. EPA's Innovative Technologies section publications website contains references to a plethora of technical documents to guide remediation[13].

Effectiveness of Spill Management

Effectiveness of remediating the legacy of toxic waste sites is often difficult to determine. In many cases when historically contaminated sites are remediated the process is only partially effective (NRC 2007). For example, depending on the on-site cleanup requirements and methods, some sites are "capped," the contaminants are left in place but the exposure pathway to environmental receptors is eliminated (e.g., Breems et al. 2009). However, in some cases a gradual leaching of contaminants into local groundwater (e.g., Wong et al. 1997) or surface water may occur that could result in releases local estuarine or marine waters.

The NRC Committee on Sediment Dredging at Superfund Megasites (2007) defined dredging effectiveness as the achievement of cleanup goals defined for each site, which take the form of remedial-action objectives, remediation goals, and cleanup levels. They also presented a framework to facilitate the evaluation of effectiveness of environmental-dredging projects at contaminated sediment sites. Their review found that evidence for dredging projects leading to achievement of long-term remedial action objectives, and within expected or projected time frames, is generally lacking (NRC 2007).The NRC Water Science and Technology Board (1988) also examined the criteria for achieving certain degrees of water quality in the areas of cleanup sites.

Physical Habitat Protection and Restoration Strategies

Protecting and restoring the physical integrity and ecological functionality of Puget Sound habitats provides the physical, chemical, and biological templates necessary for healthy fish and wildlife populations, as well as natural coastal ecosystems for human benefit. There are a variety of physical habitat protection and restoration strategies that can be applied to Puget Sound subtidal, intertidal, and shoreline marine and estuarine habitats. PSNERP has identified 21 management measures for implementing nearshore ecosystem restoration recognizing that (1) the measures can be capital projects, regulation,incentives, or education and outreach, and (2) the measures contribute to ecosystem recovery viaprotection, restoration, rehabilitation and substitution/creation (Clancy et al. 2009).These habitat measures can generally be divided into protection and restoration strategies.

Protection strategies

A key group of strategies includes the variety of regulatory and private activities that tend to protect habitats from degradation or to allow them to naturally recover their ecological function (Clancy et al. 2009). Although the initial costs of protection can be high, once the habitats are protected, the ongoing maintenance costs are often relatively low. Therefore, it is often preferable to protect currently functioning ecosystems or to protect somewhat degraded ecosystems from further degradation, allowing them recover. In some cases, it will be preferable to combine protection of the created habitats with restoration measures to speed recovery (Clancy et al. 2009).

Marine and estuarine shorelines and intertidal protection

There are a number of federal, state, tribal, local, and private programs designed to permanently protect estuarine and marine shoreline and intertidal habitats (subtidal marine protected areas are addressed below). These programs are increasingly being applied around the margins of Puget Sound. This strategy addresses PSNERP Management Measure 15, "Property Acquisition and Conservation" (Clancy et al. 2009). The PSP lists the protection of intact lands and resources as a strategic priority in the Action Agenda for Puget Sound (PSP 2008). The Puget Sound Salmon Recovery Plan (Shared Strategy 2007) highlights the importance of permanently protecting existing physical habitat as a key strategy for recovering Puget Sound Chinook (Clancy 2009).

The goal of marine and estuarine shorelines and intertidal protection is to preserve the ecological integrity of shoreline and intertidal habitats for the benefit of fish and wildlife species. It is likely that the highest functioning coastal and intertidal preserves will blend a variety of habitats, from upland forests through scrub or brush patches, beaches and/or rocky coastlines, and into the intertidal zone. At times, this transition distance may be relatively short when it occurs on steeper slopes or it may be much longer if the upland topography is relatively flat and/or the intertidal zone is broad. Clancy et al. (2009) review the variety of acquisition and protection processes as well as the various types of land and resource preservation. They also list the benefits and opportunities created by property acquisition and conservation.

Some of the metrics that could be used to decide what areas should be protected and, subsequently how well they are functioning as protected areas, include: the relative importance or critical nature of habitat types, reserve size, connectivity of migratory corridor, reducing threats to restored areas, andprotecting rare or sensitive species; Several specific property acquisition, protection, and conservation programs are explored further below.

National Estuary Program

The National Estuary Program (NEP), which was established by Congress in 1987 in amendments to the Clean Water Act. Its primary objective is to protect estuaries of national significance that are threatened by degradation caused by human activity. The program is administered by the US Environmental Protection Agency which provides funding and technical support to local NEPs. Local NEPs must be collaborative, locally driven entities that address the complex and competing issues facing the water body[14] .

Puget Sound is one of 28 nationally recognized estuaries in the NEP. The PSP Action Agenda is recognized by the NEP as programmatically focused on the same goals for Puget Sound as the NEP is. This EPA program is an important vehicle for federal funding to implement the PSP Action Agenda.

Estuarine reserves

The National Estuarine Research Reserve (NERR) program is designed to provide some level of preservation and protection to local estuaries of significance. There is one NERR in Puget Sound at Padilla Bay in Skagit County, encompassing over 11,000 acres of tidelands and marshlands. The Padilla Bay NERR[15] is managed cooperatively by the Washington Department of Ecology and NOAA. While most of the reserve is given sufficient protection to maintain ecological integrity, the NERR does not necessarily provide full protection and preservation, since many non-destructive uses are allowed, as governed by applicable state and federal laws. The various levels of protection are described in the Padilla Bay NERR management plan (Padilla Bay NERR 2008).

Beyond functionally protecting the designated estuary, Reserve staff work with local communities and regional groups to address natural resource management issues, such as non-point source pollution, habitat restoration and invasive species. Through integrated research and education, the reserves help communities develop strategies to deal successfully with these coastal resource issues.Guidance for the possible creation of additional NERRs[16] in Puget Sound can be found in several references (e.g., Kennish 2004,).

Regulations for protecting biological integrity

Government agencies and jurisdictions have implemented a plethora of laws, regulations, and guidelines designed to protect natural habitats along Puget Sound shorelines and estuaries. These regulations are targeted at both public and private lands. On private lands the regulations are designed to control the overuse or abuse of natural habitats. They address bulkheads, dredging, filling, docks, and beaches[17].

Programs for shoreline adoption, clean up, habitat enhancement and monitoring by citizen groups

Local and regional citizen volunteer groups have created programs for volunteers to help cleanup and maintain Puget Sound shorelines. For example, the Puget Soundkeepers Alliance regularly organizes clean-up days[18].

Effectiveness and certainty of estuarine and shoreline protection

Programs and regulatory processes that preserve, protect, and limit access to natural coastal habitats are considered to be among the best possible protection and restoration strategies. This is because they protect the best habitat at what is perceived to be a lower-cost than what it would cost to restore habitat after its damaged (Clancy et al. 2009). However, there apparently is little specific research on the relative effectiveness of habitat protection as compared to restoration.

This may be partly because the scientific community generally assumes that undisturbed ecosystems are automatically preferable to alter or restored habitats. Interestingly, there are many examples of species taking advantage of altered habitats such as the explosion of Caspian terns nesting on dredge spoil islands near the mouth of the Columbia River (e.g., USFWS 2005).

With the recent focus on ecosystem-based management of natural resources, there has been an upswing in research attempting to substantiate the connection between healthy critical habitats and species success. For example, several recent papers have explored the connection between the size and critical nature of habitat and the production of the species (e.g., Langton et al. 1996, Langton and Auster 1999).

Marine-protected subtidal areas

Marine protected areas (MPAs) have been applied in various settings around the world to either permanently protect critical and sensitive habitats or to temporarily allow habitat and faunal recovery from over-use. Implementation of MPAs has been viewed as a precautionary management strategy that protects functional attributes of marine ecosystems (Murray et al. 1999). Washington has 127 MPAs managed by eleven federal, state, and local agencies. These sites occur in Puget Sound and on the outer coast and cover approximately 644,000 acres and over six million feet of shoreline (Van Cleve et al. 2009). Twenty-six percent of the state's marine waters and 27% of the state's shorelines are included in the boundaries of MPAs (Van Cleve et al. 2009). The locations of many Puget Sound MPAs are shown at http://wdfw.wa.gov/fish/mpa/puget_sound/index.htm. Interested parties can also access GIS coverage layers of MPAs at http://wdfw.wa.gov/fish/mpa/puget_sound/gis_data.htm. There are also many other de-facto MPAs, such as in marine state parks, Department of Natural Resources submerged aquatic lands, etc. See also http://mpa.gov/helpful_resources/states/washington.html, for helpful links to Washington MPAs.Other resources include:

Marine Protected Areas in Washington: Recommendations of the Marine Protected Areas Work Group to the Washington State Legislature http://wdfw.wa.gov/publications/pub.php?id=00038
Marine Protected Areas in the Puget Sound Basin A tool for managing the ecosystem
http://www.vetmed.ucdavis.edu/whc/seadoc/pdfs/gaydosetal_05_mpas.pdf

MPAs are variously applied with a range of restrictions, from full protection in some MPAs, to limitations of certain activities in others. These protective measures have been demonstrated to provide excellent benefits by protecting natural areas from destructive overuse and for promoting recovery of damaged benthic habitats. They also support recovery of sessile demersal species or infauna, as well as benthic and demersal territorial fish species. For example, Halpern (2003) found in a review of 89 studies on MPAs that almost all biological metrics improved inside reserves, either compared to before reserve establishment or in comparison to similar areas outside the reserves. It must be noted, however, that whether perceived degradation of marine ecosystems can be reversed via establishment of an MPA may depend on the timescale of interest, and on whether fundamental new ecological processes have taken hold after a disturbance ends (Palumbi et al. 2008).

While improvements have been clearly observed within reserve boundaries (Halpern et al. 2003), the potential effects of reserves increasing dispersal of juveniles and adults to areas outside the reserves are less clear. Modeling results suggest that reserve networks may have the potential to enhance fishery yields under a surprisingly large number of circumstances (Gaylord et al. 2005). In at least one specific study, local dispersion and retention of molluscan shellfish larvae within and near a reserve network enhanced recruitment to local fisheries, although the effects were spatially explicit (Cudney-Bueno et al. 2009). In an Alaskan study of ling-cod, field results supported models indicating that populations increased within reserves and those populations supported increased recruitment to nearby fishing areas (Starr et al. 2004).

Scientific debate has ensued over whether MPAs, as a fishery management tool, result in improved fishery production compared to traditional methods. This is seen to depend largely on the interplay between the 1) target species, 2) nature of the larval, juvenile, and adult dispersal patterns, 3) the longevity and age at first spawning, 4) population abundance structure, 4) size of the reserve, 5) interactions between differentially affected taxa and 6) the length of time the reserve is imposed (Halpern et al. 2003, Botsford et al. 2003, Starr et al. 2004, Ruckelshaus et al. 2009).There are implications that traditional management techniques, such as size limits, seasons, and bag limits, are only partially effective at managing slow growing, late maturing, and territorial species such as rockfish and lingcod (Palsson 2001). Allison et al. (1998) concluded that MPAs were most effective when combined with other, more traditional management tools.

Application in Puget Sound

Several reviews have been done on the extent and implementation of MPAs in Puget Sound (Murray and Ferguson 1998, Palsson 2001, Van Cleve et al. 2009), but none of these are scientifically rigorous studies of their effectiveness. MPAs have been shown to be effective in certain other areas (e.g., Halpern et al 2003) and appear, at least preliminarily, to be effective in Puget Sound. The oldest Puget Sound MPA was established at Edmonds in 1970 and, as of 2001, had 15 times as many copper rockfish, as comparable nearby fished areas (Palsson 2001). Lingcod were also twice as abundant and were 50% bigger on average, than at nearby fished sites (Palsson 2001). Even if reserves are relatively small, they can still have benefits to areas outside of the reserve boundaries. For example, lingcod nests were 3 times as abundant in one Puget Sound MPA than in surrounding fished areas (Palsson 2001). The higher production of the MPA creates a dispersal mechanism to surrounding harvest areas.

Other needs for the best application of MPAs include the incorporation of fishing behavior, such as fishing just outside the reserve boundary (Kellner et al. 2007), into the management scheme that includes MPAs, as well as considerations of vertical zoning in application of MPAs (Grober-Dunsmore et al. 2008). Ultimately, the optimal management schemes, at least for fisheries management, will likely include some combination of MPAs and other management practices (Allison et al. 1998).

Potential Effectiveness of Marine Protected Areas

MPAs embrace the fundamentals of ecosystem-based management by protecting ecosystems or portions thereof (Ruckelshaus et al. 2008). There are numerous scientific analyses of MPA performance to generally support their use in habitat and species protection and restoration (e.g., Halpern et al. 2003), but specific selection, design, and implementation policies should be customized for each situation (Botsford et al. 2003, Roberts et al. 2003). If ecological, social, and economic criteria (Roberts et al. 2003) and potential resilience against climate change (McLeod et al. 2008) are carefully considered for selecting MPAs, they can be viewed as powerful tools among other marine protection and restoration strategies. A significant caveat is that much additional research is needed in both understanding the performance of specific MPAs, relative to their intended biological and/or management outcomes (e.g., White 2009), and in techniques to analyze and predict MPA performance (Pelletier et al. 2008).

Marine Spatial Planning

Marine spatial planning (MSP) is an emerging protection and restoration strategy in that it is a proactive approach for deciding which activities should have priority in certain areas and which activities are compatible or incompatible. Managing human activities to enhance compatible uses and reduce conflicts among uses, as well as to reduce conflicts between human activities and nature, are important outcomes of MSP (Ehler and Douvere 2009). It therefore encompasses decisions about the application of estuarine reserves and MPAs, described above, as well as other marine and estuarine protection, restoration, and development activities.

Well-conducted marine spatial planning can reduce conflicts between users and increase regulatory efficiency, facilitate the development of emerging industries such as wind and wave energy and aquaculture and help maintain ecological processes and the ecosystem services they support (such as fishing, marine tourism and recreation, and cultural uses of the ocean)[19].

Coastal and Marine Spatial Planning (CMSP) is a hallmark of President Obama's Executive Order on a U.S. Ocean Policy (CEQ 2010) and by a number of state, federal, and international marine planning organizations (e.g., Young et al. 2007, Ehler and Douvere 2009a,b, Commonwealth of Massachusetts 2009). The attractiveness of MSP is that it features place-based, integrated management of the full suite of human activities occurring in spatially demarcated areas identified through a procedure that takes into account biophysical, socioeconomic, and jurisdictional considerations (Young et al. 2007).

MSP Application in Puget Sound

MSP has not yet been fully applied in Puget Sound, although one of the action items in the Puget Sound Action Agenda is to "…conduct spatial (mapped) analyses to evaluate current ecosystem status and the primary threats and drivers affecting ecosystem health. Together with models and refined indicators, this work will highlight the location and relative importance of threats and drivers across the entire ecosystem, and help identify the features of Puget Sound that are most at risk" (PSP 2009). While this is not MSP in the fullest sense, this action item will establish a baseline for MSP in Puget Sound. So far, MSP in Puget Sound has occurred through site-by-site planning such as where to locate MPAs or the reservation of certain areas for industrial use or shipping lanes, etc. There is an apparent lack of a specific program aimed at implementing MSP in Puget Sound. There are a number of good models for administratively or legislatively directed MSP programs. For example, Massachusetts has been a leader in implementing a state Ocean

Management Plan (Commonwealth of Massachusetts 2009). Another overarching MSP guidance source is the step-by-step guide for implementing MSP (see Intergovernmental Oceanographic Commission 2009).

MSP is a promising strategy for the future health of Puget Sound. Just as in land-use planning, a coordinated, concerted effort to assess and allocate marine and estuarine areas for their optimal use, while protecting the ecological attributes of the Sound. Many of the components and strategies that will support MSP in Puget Sound have been, or are being, organized, such as the Puget Sound Regional Synthesis Model (PRISM)[20], the Puget Sound Ecosystem Portfolio Model (PSEPM)[21] , and the for Puget Sound Marine Environmental Modeling (PSMEM)[22] . While these tools have the potential to support MSP in Puget Sound, they are not yet specifically aimed at MSP.

In addition to the Puget Sound-specific spatial models mentioned above, many specific tools have been developed that can aid the MSP effort in Puget Sound.

MSP planning tools

MSP is an essential strategy for restoring and maintaining a healthy Puget Sound (Handbook item). There are a large number of MSP-specific planning tools already available[23,24]. There are also ecological, social, and economic criteria for selecting MPAs (Roberts et al. 2003) that can be incorporated into MSP.

Recently, more attention is being paid to the effects of vertical zoning in MPAs (Grober-Dunsmore et al. 2008), but little specific research has been accomplished on this topic. Connectivity is an important planning goal from an ecological perspective for MSP – see Australian CONNIE at http://www.per.marine.csiro.au/aus-connie/quickGuide.html Further, when planning for various uses, it is important to account for "edge" effects of users, such as the phenomenon of fishers "fishing the line" along marine reserve boundaries (Kellner et al. 2007).

Ecosystem analysis tools

A number of other ecosystem evaluation and planning tools could also be relevant as aids to MSP. See also http://code.env.duke.edu/projects/mget/wiki.

http://fishbase.sinica.edu.tw/report/t/home.htm

http://www.ecopath.org/

http://www.csiro.au/science/ps3i4.html

Integrated Ecosystem Assessment model (Levin et al. 2008)
http://www.nwfsc.noaa.gov/assets/25/6801_07302008_144647_IEA_TM92Final.pdf

Potential Effectiveness of Marine Spatial Planning

It will be somewhat difficult to asses the effectiveness or degree of uncertainty in the MSP process and, to date, there are no formal processes available for assessment of MSP uncertainty. Belfiore et al. (2006) and Ehler and Douvre (2009) outline a proposed processes for determining MSP effectiveness via establishing and monitoring indicators. Because MSP is a policy-oriented planning process, rather than a specific, physical protection or restoration strategy itself, it is less scientifically rigorous and does not easily lend itself to assessments of certainty in its outcomes. Nonetheless, MSP clearly should be included in any thorough review of marine and estuarine protection and restoration strategies. While the degree of certainty provided by MSP processes is presently undeterminable, the outcomes are clearly linked to correct regulatory decisions in the planning process and the variation in environmental conditions, enforcement of the resultant regulations, marine accidents and spills, etc. Ultimately, indicators are needed to monitor progress of MSP with respect to inputs, activities, outputs, and outcomes. Progress needs to be monitored at all levels of the system to provide feedback on areas of success, as well as areas where improvements may be needed (Belfiore et al. 2006, Ehler and Douvere 2009, Foley et al. 2010).

Ultimately, the evaluation of MSP effectiveness will be determined by whether the Puget Sound ecosystem recovers its basic dynamic ecological functionality, resiliency, and healthy fish and wildlife populations. Recovery potential and/or resistance can differ from place to place within the same marine or intertidal ecosystem (Palumbi et al. 2008). Determining effectiveness will depend on rigorous monitoring programs. Previous analyses of restoration programs have found, by studying such ecological features as species redundancy and complementarity, that recovery, resistance, and reversibility are key components of resilience (Palumbi et al. 2008). Monitoring effectiveness of marine planning has also revealed that the intended ecosystem effects of management plans are not always realized and, in fact, sometimes opposite outcomes are observed (e.g., Pine et al. 2009). There is also a critical lack of modeling tools for evaluating ecosystem-based policies (Pine et al 2009).

Marine and estuarine habitat restoration strategies

A large emphasis of Puget Sound protection and restoration strategies has been placed on physical habitat restoration. Here we discuss the variety of strategies for restoring the physical and ecological function of marine, estuarine, subtidal, intertidal, and shoreline function many of which can be expanded upon in future versions of the PSSU1. Much of the naturally occurring physical habitat in and around Puget Sound has been altered by the variety of human activities. These include diking, dredging, filling, water flow control, bulkheads, jetties, docks, bank hardening, loss of large and small estuaries, blockage of some coastal embayments, shoreline shortening, loss of natural sediment, increased unnatural sedimentation and cumulative effects of all these, as described in the chapter on threats. Shipman et al. (2008) illustrated the goal of some aspects of Puget Sound ecosystem functional restoration.

The goal of physical habitat restoration strategies is to restore connectivity and size of large river deltas, restore sediment input, transport and accretion, enhance shoreline complexity, and enhance habitat heterogeneity and connectivity. The strategies in this section speak strongly to the PSP priority B "Restore ecosystem processes, structures, and functions" and many of the Action Agenda items under that priority (PSP 2009).

Estuarine-specific habitat restoration

There are a number of documents designed to guide creation, restoration, and enhancement of coastal wetlands (e.g., Interagency Working Group on Wetlands, undated). See http://pugetsoundnearshore.org/esrp/esrp_report08.pdf.

Opening dikes and levees to recreate intertidal wetlands

This is Clancy et al. (2009) management measure 3, "Berm or Dike Removal or Modification". The strategy applies to wetlands that have been closed off by levees, dikes, and channelization. It also is relevant to pocket wetlands along natural shorelines that have been closed off by modifications of beach structure, for habitat details see Shipman et al. 2008.

Eliminating migrational barriers: Hydraulic Modification

(Clancy et al. (2009) management measure 9. This strategy involves opening culverts, tide-gates, or breachways in existing dikes and levees. Hydraulic modification allows water to flow in and out of estuarine areas more naturally and creates opportunities to reduce migrational barriers.

Physical Exclusion

The purpose of physical exclusion is to close recovering natural habitats to human access to speed the recovery process. Physical exclusion applies to beach and shoreline restoration as well as estuarine restoration, but will only be described here.

Topography restoration

Applies to both estuarine and shoreline restoration.

Includes removing hard surfaces and restore natural features at the land/water interface.

Shoreline restoration strategies

This strategy is about restoring beach and coastline function from the effects of armoring, bulkheads, docks, uplands modification, light, noise, and other longshore migrational barriers.See articles in files at PS Gen/shorelines/. Also – from J Lombard 3-1-10: WDFW has posted a new science paper, Protection of Marine Riparian Functions in Puget Sound, Washington: http://wdfw.wa.gov/hab/ahg/riparian_protection.htm. This document was developed to provide shoreline planners and managers with a summary of current science and management recommendations to inform protection of ecological functions of marine riparian areas. It was prepared by Washington Sea Grant for WDFW, with Ecology's participation and AHG review.)Clancy et al. (2009)also provide an excellent listing of the kinds of restoration activities that apply to shorelines and beaches. The strategies listed below, primarily from their list of restoration measures.

Armor Removal or Modification

- Beach Nourishment
- Debris Removal (MM 6)
- Groin Removal and Modification
- Overwater Structure Removal or Modification
- Substrate Modification

Evaluating the effectiveness of physical restoration

Assessing the scientific basis for estuarine, shoreline, intertidal, and subtidal habitat restoration effectiveness is an emerging science. There are several key manuals and guides for "how to" conduct habitat restoration (e.g., Interagency Working Group on Wetlands, undated; NRC 2001, Clancy et al. 2009). However, because extensive habitat restoration has only recently been underway, there are a few long-term, rigorous scientific evaluations of estuarine and shoreline habitat restoration effectiveness.

Some recent scientific work has been targeted at evaluating cumulative ecosystem response to restoration projects (Diefenderfer, et al. 2004, 2009). Thom (2000) noted that it is very common for aquatic ecosystem restoration projects not to meet their goals. Other papers on evaluating restoration:

Thom et al. (2205) addressed uncertainty in coastal restoration projects. They found, for example, that all of the potential sources of error can be addressed to a certain degree through adaptive management.

Submergent restoration strategies

- Eelgrass and forage fish spawning area restoration
- Artificial underwater structures
- Derelict fishing gear removal and recycling
- Reducing the effects of boat and ship traffic, military activity, and other industrial activity on Puget Sound biota
- Reducing underwater noise in the Puget Sound

Footnotes:

[1] Future versions of the PSSU can expand upon topics such as heavy metal sludge disposal, water reclamation, channel rehabilitation or creation, large wood replacement, physical exclusion, revegetation, species habitat enhancement, topography restoration, armor removal, beach nourishment, debris removal, groin removal or modification, overwater structure removal or modification, substrate modification, eelgrass and forage fish spawning area restoration, artifical

underwater structures, derelict fishing gear removal and recycling and reducing the effects of boat and ship traffic, military activity and other industrial activity on Puget Sound biota.

[2] see http://www.ecy.wa.gov/programs/eap/mar_wat/focused_southdata.html and http://www.hoodcanal.washington.edu/ for more information

[3] See http://www.kingcounty.gov/environment/wastewater/ReclaimedWater.aspx for more information

[4] see http://www.ecy.wa.gov/programs/spills/prevention/prevention_section.htm) and response actions are coordinated with the US Coast Guard (see http://www.uscg.mil/ccs/npfc/About_NPFC/opa.asp

[5] see http://www.fema.gov/emergency/nims/index.shtm

[6] http://www.nrc.uscg.mil/nrchp.html

[7] see http://www.ecy.wa.gov/programs/spills/spills.html for more details

[8] from http://www.ecy.wa.gov/programs/spills/preparedness/preparedness_section.htm

[9] http://www.ecy.wa.gov/programs/tcp/cu_support/cu_process__steps_defns.htm

[10] http://www.ecy.wa.gov/programs/tcp/sites/sites_information.html

[11] http://www.ecy.wa.gov/programs/tcp/cu_support/cu_process__steps_defns.htm

[12] http://www.epa.gov/brownfields/

[13] http://www.epa.gov/tio/pubitech.htm

[14] http://yosemite.epa.gov/r10/ECOCOMM.NSF/Watershed+Collaboration/NEP

[15] http://padillabay.gov/

[16] http://nerrs.noaa.gov/BGDefault.aspx?ID=66

[17] http://www.ecy.wa.gov/programs/sea/pugetsound/laws/center.html

[18] http://pugetsoundkeeper.org/programs/partnerships/waterway-cleanups/waterway-cleanups

[19] http://www.ebmtools.org/msptools.html

[20] http://www.prism.washington.edu/index.jsp

[21] http://geography.wr.usgs.gov/science/pugetPM.html

[22] http://www.nopp.org/nopp/project-reports/reports/06kawase.pdf

[23] http://www.ebmtools.org/msptools.html

[24] http://www.ebmtools.org/about_ebm_tools.html

References

Albertson, S. L., K. Erickson, J.A. Newton, G. Pelletier, R.A. Reynolds, and M. Roberts. 2002. South Puget Sound water quality study. Phase 1. Washington Department of Ecology, Environmental Assessment Program, Olympia, Washington. http://www.ecy.wa.gov/pubs/0203021.pdf.

Allison, G. W., Lubchenco, J., and Carr, M. H. 1998. Marine reserves are necessary but not sufficient for marine conservation. Ecological Applications 8: S79-S92.

API, NOAA, USCG, and EPA . 2005. Characteristics of Response Strategies: A Guide for Spill Response Planning in Marine Environments. American Petroleum Institute; National Oceanic and Atmospheric Administration; U.S. Coast Guard; U.S. Envirionmental Protection Agency.

Baumgartner1, D. J., W.E. Frick, and P.J.W. Roberts. 1994. Dilution models for effluent discharges (Third Edition). U.S. Environmental Protection Agency, Standards and Applied Science Division, Office of Science and Technology, Oceans and Coastal Protection Division, Office of Wetlands, Oceans, and Watersheds, Pacific Ecosystems Branch, Newport, Oregon.

Belfiore, S., J. Barbičre, R. Bowen, B. Cicin-Sain, C. Ehler, C. Mageau, D. McDougall, and R. Siron. 2006. A handbook for measuring the progress and outcomes of integrated coastal and ocean management. Intergovernmental Oceanographic Commission, Manuals and Guides, 46; ICAM Dossier, 2.

Botsford, L. W., F. Micheli, and A. Hastings . 2003. Principals for the design of marine reserves. Ecological Applications 13: S25-S31.

Boulding, R. J., editor. 1996. EPA Environmental Engineering Sourcebook. Ann Arbor Press.

Breems, J., S.Wyllie-Echeverria, E. Grossman, and J. Elliott. 2009. Thatcher Bay, Washington, Nearshore Restoration Assessment .U.S. Geological Survey, Open-File Report 2008-1369.

Council on Environmental Quality. 2010. Interagency Ocean Policy Task Force. Final recommendations of the interagency ocean policy task force. White House Council on Environmental Quality

Clancy, M. and others. 2009. Management Measures for Protecting the Puget Sound Nearshore. Washington Department of Fish and Wildlife, Puget Sound Nearshore Ecosystem Restoration Project Report No. 2009-01, Olympia, Washington. http://www.pugetsoundnearshore.org/technical_papers/mangement_measurest.pdf

Commonwealth of Massachusetts. 2009. Massachusetts Ocean Management Plan. Draft Report. http://commpres.env.state.ma.us/mop/draft_plan/v1/draft-v1-complete.pdf.

Cormack, D., editor. 1999. Response to Marine Oil Pollution - Review and Assessment, Kluwer Academic Publishers. Dordrecht, The Netherlands.

Cudney-Bueno, R., M.F. Lavín, S.G. Marinone, P.T. Raimondi, and W.W. Shaw. 2009. Rapid Effects of Marine Reserves via Larval Dispersal . PLoS ONE 4: e4140. http://www.ncbi.nlm.nih.gov/pmc/articles/PMC2612740/ Diefenderfer H., K Sobocinski, R Thom, C May, A Borde, S Southard, J Vavrinec, I., and N Sather. 2009. Multiscale Analysis of Restoration Priorities for Marine Shoreline Planning. Environmental Management 44.

Diefenderfer, H. L., G.C. Roegner, R.M. Thom, E.M. Dawley, A.H. Whiting, G.E. Johnson, K.L. Sobocinski, M.G. Anderson, and B.D. Ebberts. 2005. Evaluating cumulative ecosystem response to restoration projects in the Columbia River Estuary, Annual Report 2004. Pacific Northwest National Laboratory, PNNL-15102., Richland, WA.

Douvere, F. and C. Ehler. 2009. Ecosystem-based Marine Spatial Management: An Evolving Paradigm for the Management of Coastal and Marine Places. Ocean Yearbook 23: 1-26. http://www.unesco-ioc-marinesp.be/uploads/documentenbank/11479fdbaca3efc99f3a3d2bc77ea2b7.pdf

Ehler , C. and F. Douvere. 2009. Marine Spatial Planning: A Step-by-Step Approach toward Ecosystem-based Management. UNESCO, Intergovernmental Oceanographic Commission, Manual and Guides No. 53, ICAM Dossier No. 6. http://www.unesco-ioc-marinesp.be/uploads/documentenbank/d87c0c421da4593fd93bbee1898e1d51.pdf.

EnviroVision Corp., Herrera Environmental Consultants, Inc., and Washington State Department of Ecology. 2008. Control of Toxic Chemicals in Puget Sound, Phase 2: Improved Estimates of Loadings from Dischargers of Municipal and Industrial Wastewater. Washington State Department of Ecology, Water Quality Program, Publication Number 08-10-089, Olympia, Washington.

Etkin, D. S. 2009. Oil Spill Risk in Industry Sectors Regulated by Washington State Department of Ecology Spills Program .Environmental Research Consulting, Cortlandt Manor, NY.http://www.ecy.wa.gov/programs/spills/studies_reports/ERC%20Ecology%20Oil%20Spill%20Risk%20Analysis-Final.pdf.

Foley, M. M., B. S. Halpern, F. Micheli, M. H. Armsby, M. R. Caldwell, C. M. Crain, E. Prahler, N. Rohr, D. Sivas, M. W. Beck, M. H. Carr, L. B. Crowder, J. Emmett Duffy, S. D. Hacker, K. L. McLeod, S. R. Palumbi, C. H. Peterson, H. M. Regan, M. H. Ruckelshaus, P. A. Sandifer, and R. S. Steneck. 2010. Guiding ecological principles for marine spatial planning. Marine Policy In Press, Corrected Proof.

Fresh, K. C. Simenstad J. Brennan M. Dethier G. Gelfenbaum F. Goetz M. Logsdon D. Myers T. Mumford J. Newton H. Shipman C. Tanner. 2004. Guidance for protection and restoration of the nearshore ecosystems of Puget Sound. Washington Sea Grant Program, University of Washington, Puget Sound Nearshore Partnership Report No. 2004-02, Seattle, Washington.

Frick, W. E. 2003. Visual Plumes mixing zone modeling software. Environmental Modeling & Software 19: 645-654.

Gaylord B, S.D. Gaines, D.A. Siegel, and M.H. Carr. 2005. Marine reserves exploit population structure and life history in potentially improving fisheries yields. Ecological Applications. 15: 2180-2191.

Goetz, F., C. Tanner, C.S. Simenstad, K. Fresh, T. Mumford, and and M. Logsdon. 2004. Guiding restoration principles. Washington Sea Grant Program, University of Washington, Puget Sound Nearshore Partnership Report No. 2004-03. Seattle, Washington. http://www.pugetsoundnearshore.org/technical_papers/lessons.pdf.

Grober-Dunsmore, R., L.Wooninck, J. Field, C. Ainsworth, J. Beets, S. Berkely, J. Bohnsack, R. Boulon, R., Brodeur, J. Brodziak, L. Crowder, D. Gleason, M. Hixon, L. Kaufman, B. Lindberg, M. Miller, L., Morgan, and C. Wahle. 2008. Vertical Zoning in Marine Protected Areas: Ecological Considerations for Balancing Pelagic Fishing with Conservation of Benthic Communities. Fisheries 33: 598-610.

Halpern, B. 2003. The impact of marine reserves: Do reserves work and does reserve size matter? Ecological Applications 33: 117-137.

Hart Crowser, Inc., Washington State Department of Ecology, United States Environmental Protection Agency, and Puget Sound Partnership. 2007. Phase 1: Initial Estimate of Toxic Chemical Loadings to Puget Sound. Washington State Department of Ecology; Publication Number 07-10-079, Olympia, WA.http://www.ecy.wa.gov/pubs/0710079.pdf.

Interagency Workgroup on Wetland Restoration. Undated. An introduction and user's guide to wetland restoration, creation, and enhancement. National Oceanic and Atmospheric Administration, Environmental Protection Agency, Army Corps of Engineers, Fish and Wildlife Service, and Natural Resources Conservation Service, Silver Spring, Maryland.

Jeansonne, J., Editor. 2007. An FOSC's Guide to NOAA Scientific Support. NOAA's Office of Response and Restoration, Emergency Response Division (ERD) .http://response.restoration.noaa.gov/book_shelf/1395_FOSC_Guide.pdf.

Kellner, J. B., I. Tetreault, S.D. Gaines, and R.M. Nisbet . 2007. Fishing the line near marine reserves in single and multi-species fisheries. Ecological Applications 17: 1039-1054.

Kennish, M. J. 2004.Estuarine research, monitoring, and resource protection. CRC Press. Boca Raton, Florida.

Konar, S. and M.A. Cohen. 1997. Information as Regulation: The Effect of Community Right to Know Laws on Toxic Emissions. Journal of Environmental Economics and Management 32: 109-124.

Lane, P., editor. 1995. The use of chemicals in oil spill response. American Society for Testing of Materials. Philadelphia.

Langton, R. W. and Auster, P. J. 1999. Managing essential fish habitat: What are the next steps? Fisheries 24: 30-31.

Langton, R. W., R.S. Steneck, V. Gotceitas, F. Juanes, and P. Lawton. 1996. The Interface between Fisheries Research and Habitat Management. North American Journal of Fisheries Management 16: 1-7.

Lehr, J. H., M. Hyman, T. Gass, and W.J. Seevers. 2001. Handbook of complex environmental remediation problems, ed. McGraw-Hill. New York, NY.

Levin, P. S., R. W. Zabel, and J. G. Williams. 2001. The road to extinction is paved with good intentions: negative effects of fish hatcheries on threatened salmon. Proceedings of the Royal Society of London. Series B, 268: 1153-1158. Lombard, J. 2006. Saving Puget Sound: a conservation strategy for the 21st century. American Fisheries Society. Bethesda, Maryland.

Lubliner, B., M. Redding, and D. Ragsdale. 2010. Pharmaceuticals and Personal Care Products in Municipal Wastewater and Their Removal by Nutrient Treatment Technologies, Publication Number 10-03-004. Washington State Department of Ecology, Olympia, WA.

Malins, D. C., B.B. McCain, D.W. Brown, S. L. Chan, M.S. Myers, J.T. Landahl, P.G. Prohaska, A.J. Friedman, and L.D. Rhodes. 1984. Chemical pollutants in sediments and diseases of bottom-dwelling fish in Puget Sound, Washington. Environmental Science and Technology 18: 705-713.

McCain, B. B., D.W. Brown, M.M. Krahn, M.S. Myers, R.C. Clark Jr., S. Chan, and D.C. Malins. 1988. Marine pollution problems, North American West Coast . Aquatic Toxicology 11: 143-162.

McLeod, E., R. Salm, A. Green, and J. Almany . Designing marine protected area networks to address the impacts of climate change . Frontiers in Ecology and the Environment 7: 362-370.

Murray, M. R. and L. Ferguson. The Status of Marine Protected Areas in Puget Sound. Pages 783-793 in Proceedings of the 1998 Puget Sound Georgia Basin Research Conference.

Murray, S. N., Ambrose, R. F., Bohnsack, J. A., Botsford, L. W., Carr, M. H., Davis, G. E., Dayton, P. K., Gotshall, D., Gunderson, D. R., Hixon, M. A., Lubchenko, J., Mangel, M., MacCall, A., McArdle, D. A., Ogden, J. C., Roughgarden, J., Starr, R. M., Tegler, M. J., and Yoklavich, M. M. 1999 . No-take reserve networks: Sustaining fishery populations and marine ecosystems. Fisheries 24: 11-25.

Neuman, M., D. St. John, C. Stem, J. Knauer, and N. Salafsky. 2009. Using Results Chains to Develop Objectives and Performance Measures for the 2008 Action Agenda. Technical Memorandum, November 2009. http://www.psp.wa.gov/pm.php.

Neuman, M., D. St. John, and J. Knauer.2009. Identification, Definition and Rating of Threats to the Recovery of Puget Sound. Technical Memorandum, November 2009. http://www.psp.wa.gov/downloads/2009_tech_memos/SOS_2009_tech_memo_threats_2009-06-11_FINAL.pdf.

Neuman, M., D. St. John, S. Redman, K. Stiles, N. Salafsky, J. Knauer, and M.B. Brown. 2009. Identification of Ecosystem Components and Their Indicators and Targets. Technical Memorandum, November 2009. http://www.psp.wa.gov/downloads/2009_tech_memos/SOS_2009_Tech_Memo_components_2009_06_11_FINAL.pdf.

Newton, J., C. Bassin, A. Devol, M. Kawase, W. Ruef, M. Warner, D. Hannafious, and R. Rose. 2007. Hypoxia in Hood Canal: An overview of status and contributing factors. Upl. Ms. Proceedings of the Georgia Basin Puget Sound Rseaecrh Conference. http://www.engr.washington.edu/epp/psgb/2007psgb/2007proceedings/papers/9a_newton_comp.pdf.

NRC (Committee on Contaminated Marine Sediments). 1995. Contaminated Sediments in Ports and Waterways: Cleanup Strategies and Technologies. The National Academies Press. Washington. D.C.

NRC (Committee on Sediment Dredging at Superfund Megasites). 2007. Sediment Dredging at Superfund Megasites: Assessing the Effectiveness. The National Academies Press. Washington. D.C.

NRC (National Research Council, Committee on Mitigating Wetland Losses). 2001. Compensating for Wetland Losses Under the Clean Water Act. NAS Press. Washington, D.C.

NRC (National Research Council). 1993. Managing Wastewater in Coastal Urban Areas, National Academy Press. Washington, D.C.

NRC (Water Science and Technology Board). 1988. Hazardous Waste Site Management: Water Quality Issues. National Academy Press. Washington. D.C.

Ornitz, B. E. and M.A. Champ. 2002. Oil Spills First Principles: Prevention and Best Response. Elsevier Ltd.

Padilla Bay NERR. 2008. Padilla Bay National Marine Estuarine Research Reserve Management Plan. Washington Department of Ecology, Publication no. 09-06-016. http://www.ecy.wa.gov/pubs/0906016.pdf.

Palsson, W. A. 2001. The Development of Criteria for Establishing and Monitoring No-take Refuges for Rockfishes and Other Rocky Habitat Fishes in Puget Sound. Puget Sound Research 2001. http://www.vetmed.ucdavis.edu/whc/seadoc/pdfs/MPApdfs/Palsson2001.pdf

Palsson, W. A. 2001. Marine refuges offer haven for Puget Sound fish. Fish & Wildlife Science. http://wdfw.wa.gov/science/articles/marine_sanctuary/

Palumbi, S. R., K.L. McLeod, and D. Grunbaum. 2008. Ecosystems in Action: Lessons from Marine Ecology about Recovery, Resistance, and Reversibility. Bioscience 58: 33-42.

Paulson, A. J., R.A.Feely, H.C. Curl Jr., E.A. Crecelius, and G.P. Romberg. 1989. Separate dissolved and particulate trace metal budgets for an estuarine system: An aid for management decisions. Environmental Pollution 57: 317-339.

Pelletier, D., C. Joachim, J. Ferraris, L. Benedetti-Cecchi, and J.A. Garcģa-Charton. 2008. Models and indicators for assessing conservation and fisheries-related effects of marine protected areas. Canadian Journal of Fisheries and Aquatic Sciences 65: 765-779.

Pine, W. E., S.J.D. Martel, C.J. Walters, and J.F. Kitchel. 2009. Counterintuitive Responses of Fish Populations to Management Actions: Some Common Causes and Implications for Predictions Based on Ecosystem Modeling. Fisheries 34: 165-180.

Prugh, L. R., C.J. Stoner, C.W. Epps, W.T. Bean, W.J. Ripple, A.S. LaLiberte, and J.S. Brashares. The Rise of the Mesopredator.Bioscience 59: 779-791.

PSP (Puget Sound Partnership). 2009. Action Agenda. Puget Sound Partnership, Olympia, Washington. http://www.psp.wa.gov/aa_action_agenda.php.

Puget Sound Action Team. 2007. 2007 Puget Sound Update: Ninth report of the Puget Sound Assessment and Monitoring Program. Puget Sound Action Team., Olympia, Washington. http://www.psp.wa.gov/downloads/SOS07/2007_PS_Update.pdf.

Puget Sound Harbor Safety Committee. 2008. Puget Sound Harbor Safety Plan. Puget Sound Harbor Safety Committee. http://www.pshsc.org/sitebuildercontent/sitebuilderfiles/harborsafetyplanfinal72909fullcolor2.pdf.

Roberts, C. M., S. Andelman, G. Branch, R.H. Bustamante, J.C. Castilla, J. Dugan, B.S. Halpern, K.D. Lafferty, H. Leslie, J. Lubchenco, D. McArdle, H.P. Possingham, M. Ruckelshaus, and R.R. Warner . 2003. Ecological criteria for evaulating candidate sites for marine reserves. Ecological Applications 13: 199-214.

Roberts, M., S. Albertson, A. Ahmed, and G. Pelletier. Upl Ms. South Puget Sound Dissolved Oxygen Study: South and Central Puget Sound Water Circulation Model Development and Calibration . Publication no. 09-03-0xxx.

Roberts, P. J. W., W. H. Snyder, and D. J. Baumgartner. 1989. Ocean Outfalls. III: Effect of Diffuser Design on Submerged Wastefield. Journal of Hydraulic Engineering, 115: 49-70.

Ruckleshaus, M., T. Klinger, N. Knowlton, and D.P. DeMaster. 2008. Marine Ecosystem-based Management in Practice: Scientific and Governance Challenges. Bioscience 58: 53-63.

Ruckelshaus, M., T. Essington and P. Levin.2009. Marine ecosystem-based management in practice: Puget Sound, Washington, USA. in McLeod and Leslie. Ecosystem-based management for the oceans. Island Press, Washington, DC

Sayari, A., S. Hamoudi, and Y.Yang. 2005. Applications of Pore-Expanded Mesoporous Silica. 1. Removal of Heavy Metal Cations and Organic Pollutants from Wastewater. Chemistry of Materials 17: 212-216.

Sellers, K. 1999.Fundamentals of Hazardous Waste Site Remediation. CRC Press. Boca Raton, FL.

Servos, M. R., D.T. Bennie, B.K. Burnison, A. Jurkovic, R. McInnis, T. Neheli, A. Schnell, P. Seto, S.A. Smyth, and T.A. Ternes . 2005. Distribution of estrogens, 17b-estradiol and estrone, in Canadian municipal wastewater treatment plants. Science of the Total Environment 336: 155-170.

Shipman, H. 2008.A Geomorphic Classification of Puget Sound Nearshore Landforms. Seattle District, U.S. Army Corps of Engineers, Seattle, Washington., Puget Sound Nearshore Partnership Report No. 2008-01.

Simenstad, C., M Logsdon, K. Fresh, H. Shipman, M. Dethier, and J. Newton . 2006. Conceptual model for assessing restoration of Puget Sound nearshore ecosystems. Washington Sea Grant Program, University of Washington, Available at , Puget Sound Nearshore Partnership Report No. 2006-03 , Seattle, Washington. http://pugetsoundnearshore.org.

Sivas, D. A. and M.R. Caldwell. 2008. A New Vision for California Ocean Governance: Comprehensive Ecosystem-Based Marine Zoning. Stanford Environmental Law Journal 27: 209-270.

SSDC (Shared Strategy Development Committee). 2007. Puget Sound Salmon Recovery Plan. Shared Strategy for Puget Sound, Plan adopted by the National Marine Fisheries Service (NMFS) Seattle, Washington. http://www.nwr.noaa.gov/Salmon-Recovery-Planning/Recovery-Domains/Puget-Sound/PS-Recovery-Plan.cfm.

Starr, R. M., V. O'Connell, and S. Ralston. 2004. Movements of lingcod (Ophiodon elongatus) in southeast Alaska: potential for increased conservation and yield from marine reserves. Canadian Journal of Fisheries and Aquatatic Sciences 61: 1083-1094.

Sumpter, J. P. and A.C. Johnson. 2005. Lessons from Endocrine Disruption and Their Application to Other Issues Concerning Trace Organics in the Aquatic Environment.

Environmental Science and Technology 39: 4321-4332. Suryan, R. M., V.S. Saba, B.P. Wallace, S.A. Hatch, M. Frederiksen, and S. Wanless. 2009. Environmental forcing on life history strategies: Evidence for multi-trophic level responses at ocean basin scales. Progress in Oceanography 81: 214-222.

Thom, R. M. 2000.Adaptive management of coastal ecosystem restoration projects. Ecological Engineering 15: 365-372.

Thom, R. M., G.D. Williams, A.B. Borde, J.A. Southard, S.L. Sargeant, D.L. Woodruff, J.C. Laufle, and S. Glasoe. 2005. Adaptively Addressing Uncertainty in Estuarine and Near Coastal Restoration Projects. Journal of Coastal Research 40: 94-108.

UNESCO. Marine Spatial Planning initiative. http://www.unesco-ioc-marinesp.be/

USFWS et al. (U.S. Fish and Wildlife Service, U.S. Army Corps of Engineers, and NOAA Fisheries). 2005. Caspian Tern Management to Reduce Predation of Juvenile Salmonids in the Columbia River Estuary: Final Environmental Impact Statement. U.S. Fish and Wildlife Service; Migratory Birds and Habitat Programs, Portland, Oregon.

Van Cleve, F. B, C.Simenstad, F. Goetz, and T.Mumford. 2004. Application of the "Best Available Science" in Ecosystem Restoration: Lessons Learned from Large-Scale Restoration Project Efforts in the USA. Washington Sea Grant Program, University of Washington, Puget Sound Nearshore Partnership Report No. 2004-01., Seattle, Washington. http://www.pugetsoundnearshore.org/technical_papers/lessons.pdf.

Van Cleve, F. B., G. Bargmann, M. Culver, and the MPA Work Group. 2009. Marine Protected Areas in Washington: Recommendations of the Marine Protected Areas Work Group to the WashingtonState Legislature. Washington Department of Fish and Wildlife, Olympia, WA.

Water Environment Federation (WEF). 2006. Membrane Systems for Wastewater Treatment. WEF Press, McGraw-Hill, New York, NY.

WDOE (Washington Department of Ecology). 2008. South Puget Sound Dissolved Oxygen Study Key Findings on Nitrogen Sources from the Data Report .Publication no. 08-10-099.http://www.ecy.wa.gov/pubs/0810099.pdf.

White, C. 2009. Density dependence and the economic efficacy of marine reserves.Theoretical Ecology 2: 127-138.

Wong, J., H.L.Chin, and G.L. Nolen. 1997. Design of remediation systems. Lewis Publishers. Boca Raton, FL.

Young, K. A. 2001. Defining units of conservation for intraspecific biodiversity: Reply to Dimmick et al. Conservation Biology 15: 784-787.

Fisheries and Wildlife Protection and Restoration Strategies

Editor's Note: This section is in outline form except for the Discussion of Harvest Management

1. Introduction

Salmon and steelhead protection and restoration

A. Life-history-based restoration

B. ESA restoration vs. full, optimum production

C. The 4-H approach

1. **Potential Strategies: Habitats**

a. Protection and restoration of instream habitat complexity

b. Removal of barriers (culverts, small dams, etc.)

c. Access to off-channel habitats, including intertidal estuaries

d. Normal run-off patterns and instream flows

i. Irrigation diversion by-passes and intake mortalities

ii. Excessive groundwater removals that reduce stream flows

e. Reduction of excess sediment loads from upstream areas

f. Protection of critical salmon habitats

2. **Potential Strategies for Hydropower and other major dams**

a. Upstream and downstream passage of adults and juveniles or dam removal

b. Flow operations that benefit fish

c. Manage or eliminate water withdrawals that cause instream flow reductions

3. **Potential Strategies for Hatcheries**

a. Operate hatchery programs within the context of their ecosystems

b. Operate programs as genetically integrated or completely separate stock management programs, e.g., separate the harvest of hatchery and wild fish in time space and/or by harvest method

c. Size programs consistently with their stock goals and with the carrying capacity of the freshwater and marine ecosystems

d. Ensure productive habitat for hatchery programs

e. Emphasize quality, not quantity in fish releases

f. Use in-basin rearing and locally adapted broodstocks

g. Maintain genetic integrity by spawning adults randomly throughout the run

h. Use genetically benign spawning protocols that maximize effective population size

i. Reduce risks associated with outplanting and net pen releases

j. Develop a system of wild steelhead zones

k. Use hatchery carcasses for nitrifying streams

4. **Harvest - Puget Sound Salmon Harvest Management as a Restoration Strategy**

Harvest management is one of the four "Hs" essential for the recovery of Puget Sound Salmon (Shared Strategy 2007). Harvest management is critical because it determines both the number of spawners that reach spawning habitat as well as the number of fish available for harvest.

We suggest six interrelated harvest management strategies that could be applied to Pacific salmon restoration in Puget Sound. If simultaneously implemented, they will set the stage for improved escapement of spawners to the freshwater habitat, ultimately leading to improved run sizes (assuming the other Hs are well-managed). These strategies are: 1) improved estimates of salmon carrying capacity, 2) improved run-size forecasting, 3) improved accuracy of in-season harvest management, 4) avoidance of genetic alteration of stock structure and diversity via harvest, 5) fully functional, realistic tools for harvest management decisions, and 6) monitoring of escapement and harvests.

Salmon Habitat Carrying Capacity

Understanding salmon carrying capacity is a key component of salmonid restoration. Because of chronic overfishing, habitat degradation and, more recently, habitat restoration, salmon managers have lost the baseline reference points for freshwater and estuarine production (Pauley 1995). Furthermore, awareness is increasing about the critical nature of carrying capacity in the nearshore marine and oceanic habitats (e.g., Ruggerone et al. 2003, 2005) and the relative importance of early marine survival (Farley et al. 2007, Van Doornik et al. 2007). However, without a more complete understanding of these limitations, it is difficult to assess whether restoration of salmon populations is working. Better habitat-based benchmarks are needed from which to manage the restoration process.

It is important to note that the recovery benchmarks of the Endangered Species Act (ESA) recovery plans are not necessarily the same goals for full restoration. This is because the ESA is designed to ensure that populations do not go extinct, rather than ensure that they attain their full production capacity which, when restored, will in turn support healthy aquatic ecosystems and tribal, commercial, recreational fisheries.

The science of salmon capacity estimation has only been partially developed. For decades, capacity for many salmon stocks was estimated using retrospective statistical models of the relationship between the number of spawners and the subsequent count of returning adults (Ricker 1958, 1975; Beverton-Holt 1957). While these models perform adequately under ideal conditions, they have been shown to be fraught with numerous technical weaknesses (Hilborn and Walters 1992, Knudsen 2000). More recently, the science of salmon capacity has been expanding based on observations of the numbers of juveniles produced per spawner and per habitat area and expressed in various models (e.g., EDT – Mobrand et al. 1997 , SHIRAZ – Scheuerell et al. 2006, Ripple – Dietrich and Ligon 2008, UCM – e.g., Cramer and Ackerman 2009). Research and development is expanding on life history-based models that incorporate both habitat and capacity in the relationship of salmon to their environment, as it ultimately influences survival.

Some recent progress has been made in estimating salmon carrying capacity and related modeling to support harvest management as a salmon restoration strategy. However, strategic implementation of these techniques requires further scientific advancements as suggested by Hilborn (2009) and Knudsen and Michael (2009). Some of the remaining challenges are 1) determining how many fish should be produced per habitat, 2) better ascertaining how the environment influences salmon survival and production, 3) developing functional, realistic simulations that can be used for more precise management decisions, 4) accounting for the interactive effects of habitat, hatcheries, and other salmon species, and 5) correcting for the lack of accurate and/or long-term data (by stock).

Preseason and In-Season Run Size Forecasts

Salmon restoration also depends on improving both pre-season and in-season run size forecasts so that decisions about harvest management can be tuned to the number of adults expected to return. Successful forecasting is extremely challenging because the number of returning fish depends on dynamic and complex interaction between the often unknown number of smolts entering the ocean and the highly variable ocean environment. Pre-season forecasting is important for management decisions for determining expected escapements and, by subtraction, opening or closing the various fisheries. Current run forecasting is generally relatively inaccurate. For example, Puget Sound Chinook pre-season forecasts of escapement were only within 10% of the actual escapement values for 12% of the forecasts between 2001 and 2006 (PSIT and WDFW 2008). Therefore, the current strategy for managers working to rebuild depleted runs is to set harvest rates relatively low to account for the highly variable returns (e.g., PSIT and WDFW 2009). There is increasing evidence, however, that forecasts can be improved by additional research into the relationships between salmon survival and environmental drivers (Beamish et al. 2009, Noakes and Beamish 2009). When managers have more accurate forecasts, they will be able to refine harvest management decisions.

Short-term forecasting could be improved by increasing the frequency and accuracy of in-season fisheries-dependent sampling of open fisheries, in test fisheries in closed areas, and by monitoring in-river escapement with weirs, traps, and/or sonar (e.g. Clark et al. 2006). Research is gradually revealing technical methods that will make in-season predictions more reliable, such as the ability to use coded-wire tag information to refine run predictions (e.g. Holt et al. 2009), but more research is needed.

In-Season Harvest Management

Improved precision in spatial and temporal management is a strategy to separate capture of abundant stocks, such as plentiful wild fish or those from hatcheries, from "incidentally" captured depleted or jeopardized stocks (NRC 1996, SSDC 2007). There are two major types of suggested improvements in-season harvest management: 1) techniques that make harvest management decisions more precise, and 2) harvest techniques that separate harvestable from non-harvestable fish. Clearly, if these two kinds of techniques can be improved, harvest managers will be able to more carefully determine which and how many fish may be harvested and which fish may escape to spawn (Knudsen and Doyle 2006).

The more precise the in-season harvest management, the more likely abundant stocks, such as hatchery or abundant wild fish, can be harvested without harming wild populations that can only withstand a much lower harvest rate. The major features of such a scheme include identifying the relative abundance of each stock present in each fishing area at any given time, and then opening or closing the fishing area as necessary, as described and preliminarily modeled by Newman (2000). An important component is real-time stock separation within each fishing area. Recent advancements in genetic stock identification are increasingly improving the technical ability for accuracy and precision of stock separation (e.g., Smith et al 2005, Dann et al. 2009), although these techniques have not yet been widely applied for Puget Sound stocks. Currently, stock mixtures are determined for chinook and coho from coded-wire-tag data of representative hatchery stocks. Under ideal management, as the fish move sequentially through the management areas, decisions could be made to protect weak stocks and/or allow harvest of abundant stocks(e.g., Newman 1998). For example, increasing the accuracy of in-season harvest management, has been shown to be effective in maintaining healthy Alaskan salmon runs, even though they are routinely subjected to moderate to heavy fishing (e.g., Clark et al. 2006). Further research as well as dedication to in-season sampling is needed for such real-time decision-making to be effective in Puget Sound salmon management.

Selective fishing methods and gears allow release of incidentally captured non-targeted stocks to escape unharmed.Because of a lack of external indicators of stock origin, selective salmon fisheries can only be applied to hatchery versus wild stocks by externally marking all hatchery fish (e.g., HSRG 2004, Kostow et al. 2009, McHugh et al. 2009). Some fishing gears are better suited to selective fisheries. For example, fish wheels, purse seines, and traps allow non-target fish to be released unharmed, while gill nets tend to preclude live release (e.g., Copes 2000).

Both recreationally and commercially caught salmon and steelhead can be released unharmed (e.g., Vander Haegen et al. 2004, Cowen et al. 2007), although there can be various amounts of delayed mortality and/or failures to spawn due to catch and release, or escapement from gears (e.g., Wertheimer, 1988, Baker and Schindler 2009). Additional research is needed to advance the science and management of selective fisheries.

In summary, the status of applying precision forecasting and in-season harvest techniques to Puget Sound salmon harvest management is mixed. While in some cases the techniques described above are being partially applied, in many cases additional technical advancements are needed. For example, Puget Sound Chinook salmon harvests (hence escapements) are managed primarily by setting a relatively low harvest rate (PSIT & WDFW) in part to accommodate the fact that there is insufficient information to manage the in-season run precisely.

Avoidance of Genetic Changes Due to Harvest

To sustain the productivity of harvested populations, there are important genetic considerations for harvest management which havethe potential to cause three types of genetic change: 1) alteration of population subdivision; 2) loss of genetic variation; and 3) selective genetic changes (Allendorf et al. 2008). Population subdivision occurs through changes in the metapopulation structure of Pacific salmonids that coincides with the network of natal river populations (Policansky and Magnuson 1998, Gustafson et al. 2007). This also includes the subpopulation diversity represented by the variety of intraspecific life history types. Loss of genetic variation occurs when a single population is reduced to too few spawners (Waples 1990). Allendorf et al. (2008) also suggest that, as the population size decrease, there can be a loss of fitness selection. Selective genetic changes in salmon populations can be induced by harvests. Hard et al. (2007) reported strong evidence that selection intensity and genetic variability in salmon fitness traits from fishing can cause detectable evolution within ten or fewer generations. Salmon body size and run timing are two heritable traits, among others, that have been demonstrated to be affected by fishing (e.g., Hamon et al. 2000, Quinn et al. 2002).

Allendorf et al. (2008) recommended recognizing that some genetic change due to harvest is inevitable and that harvest management plans should be developed by applying basic genetic principles combined with molecular genetic monitoring to minimize harmful genetic change. These issues need further study in Puget Sound so that harvest management plans can be refined to reduce fisheries-induced genetic selection.

Monitoring of Escapement, Harvests, and Smolts

We suggest that ideal salmon and steelhead population management consists of monitoring two population variables: adult run size and smolt production. For successful long-term harvest management and future planning, total salmon run sizes should be estimated after each season. This requires accurate monitoring of the harvest, attributed to each population, plus the escapement of each population. In Washington, harvest estimates are made via fish sales tickets for commercial harvests and by the sport catch record card system for sport harvests (SSDC 2007), each of which has inherent inaccuracies. Assignment of the harvest to the river of origin is a key component of the final estimates, the accuracy of which depends upon the type of fishery (e.g., marine harvests tend to be mixed while freshwater harvest are more likely to be assigned to the correct river of origin). Catches of chinook, coho, and chum from mixed stock fishery areas are separated post-season by some combination of coded-wire-tag (CWT) recovery data and genetic baseline information (e.g., Johnson et al. 1997, PSIT and WDFW 2008). Once estimates are assigned to their rivers of origin, cohort reconstructions enable estimation of exploitation rates, which may be compared to results from the fishery regulation and assessment model (FRAM) estimates (PSIT and WDFW 2004, PFMC 2007). This process is variably imprecise depending on the species, available data, and model used (e.g., Starr and Hilborn 1988, Johnson et al. 1997).

To obtain the total estimated run size, annual escapement of spawners are added to the estimated harvest numbers. Escapements are monitored by a variety of methods, but not all streams/populations are monitored and the data quality of those that are monitored is highly variable (Knudsen 2000). For example, there is no escapement information on summer steelhead and escapement estimates are unavailable for four winter steelhead runs (PSSTRT 2005). The Salmonid Stock Inventory (SaSI) by WDFW is a standard process for monitoring and recording the escapements or indices of escapement, also used to assess the overall status of the stocks[1]. However, it was last updated in 2002 and, at that time, there was insufficient data for 28% of known Puget Sound stocks[1]. Assessment of smolt production is also important for optimal harvest management. Having both adult and smolt metrics for a given population allows the discernment of both freshwater and marine survival. WDFW presently uses the Intensively Monitored Watershed (IMW) program to monitor smolt production for selected species (Bilby et al. 2004), including six locations in Puget Sound. The basic concept is that these watersheds represent a sampling of all watersheds and that IMW observations on smolt production may be expanded to other similar watersheds (Bilby et al. 2004). Coho smolts are also monitored in several Puget Sound streams by WDFW.

Harvest Management Tools

Currently, run forecasting tools generally consist of past years' run reconstructions, combined with observations on brood-year survival conditions, and, in some cases, observed smolt production, to estimate predicted run returns (e.g., PSIT and WDFW 2008). The primary harvest management modeling tool for Chinook and coho is FRAM (PFMC 2007). Current salmon run forecasting is a highly variable science (Adkison and Peterman 2000, Beamish et al. 2009). For example Puget Sound Chinook forecasts ranged between -403% to +88% of the subsequently observed run sizes (PSIT and WDFW 2008).

Full salmon restoration will require progress on all the topics described above plus the concomitant development of improved computer-based, decision-making tools as described by Knudsen and Michael (2009). There are a number of possible scenarios for how modeling tools could be improved, but perhaps the most the most promising approach is exemplified when the all-H analyzer (AHA) is used to evaluate the management options (e.g., Kaje et al. 2008). Inputs on habitat-based recovery goals are obtained from SHIRAZ (Scheuerell and Hilborn (2006) and/or EDT (Mobrand et al. 1997). AHA then allows the user to concurrently model alternative scenarios for habitat, harvest, and hatcheries. However, there are many opportunities for improvement to the AHA model. In regard to improving AHA, for example, habitat is modeled as a simple production curve and harvest is modeled as a fixed exploitation rate. The model could be made much more useful by incorporating a model of the relationship of life-stage-specific survivals to habitat conditions (Hilborn 2009), perhaps through modifications of SHIRAZ. This also has the advantage of being able to incorporate interactions between hatchery and wild fish at a number of life stages, as recommended by Hilborn (2009). Another necessary improvement is to incorporate more detailed harvest management modeling, such as by including key outputs of the FRAM model currently used for fishery management. This would allow evaluation of the effects of selective fisheries and/or the impacts of different fishery plans on different life history types (i.e., diversity) (Hilborn 2009). AHA is currently focused on the degradation of wild stock productivity due to the presence of hatchery fish, mainly arising from deleterious genetic effects (Michael et al. 2009). However, with incorporation of improved habitat and harvest modules, the model could also include a number of other hatchery effects that are currently ignored in AHA. Additionally, modifications to make the AHA model stochastic are needed (Hilborn 2009). Lastly, further model developmentis necessary to include the interactions of multiple species (e.g., Greene and Pess 2009).

Summary

To date, harvest management restoration strategies have included relatively unknown habitat capacity, harvest management information inaccuracies, lack of in-season management techniques, and therefore complicated negotiations, often contentious because of the uncertainty of run-sizes. Such strategies have only been partly effective. Overall, the scientific basis for harvest-related salmon restoration could be considered to be “developing” in that much of the science, especially the basic biology of the species, is reasonably advanced, but certain critical information is still lacking. At this point in the development of salmon harvest management science, we can articulate some ways to improve accuracy and precision:

- improved methods for estimating salmon and steelhead carrying capacity,
- better run-size forecasting,
- improved accuracy and precision of in-season harvest management,
- better ways to avoid genetic alteration of stock structure and diversity,
- increased monitoring of escapement, harvests, and smolts, and
- advanced tools for harvest management decisions

Outline for the rest of Section 5.2

D. Restoration strategies that integrate the 4-Hs

1. Comprehensively model fisheries populations, including all management and restoration systems

2. Fisheries management plans, including ESA recovery plans

E. Case examples of successful protection and restoration strategies

Resident freshwater fish and anadromous fish other than salmon and steelhead

A. Habitat

B. Management plans and recovery plans (for listed species)

C. Use of hatchery programs

D. Harvest management via regulations

Wildlife in the watersheds.

A. Habitat

B. Nutrients

C. Management plans

D. Harvest management

Marine fisheries protection and restoration

A. Habitat protection and restoration (as described in the next chapter)

B. Marine protected areas for improved management

C. Forage food availability and management

D. Stock rebuilding

E. Harvest management via regulations

F. Fisheries management and/or recovery planning

Shellfisheries protection and restoration

A. Intertidal and subtidal habitat protection and restoration (as described in the next chapter)

B. Harvest management via regulations

C. Stock rebuilding

D. Use of shellfish hatcheries

E. Fisheries management planning

Marine mammals -- Pinnipeds, Cetaceans, Sea Otters

A. Habitat protection and restoration (as described in Chapter 3 and in the next chapter)

B. Closed boating areas for improved management

C. Forage food availability and management

D. Stock rebuilding via federal and state management and/or recovery planning

Puget Sound birds -- Waterfowl, Shorebirds, Seabirds

A. **Habitat protection and restoration (as described in Chapter 3 and in the next chapter), but specifically:**

1. Nesting locations

2. Feeding areas

3. Resting areas

B. Forage food availability and management

C. Interspecific interactions

D. Population rebuilding via federal and state management and/or recovery planning

Invasive species

A. Establish a program to reduce and, where possible, eliminate the introduction and spread of non-native species

B. Identify and rank non-native, invasive species that cause or have the potential to cause significant negative impacts to the Puget Sound ecosystem

The effectiveness of recovery planning

Footnotes

[1] seehttp://wdfw.wa.gov/fish/sasi/index.htm

References

Adkison, K. D. K., Artru, A. A., Powers, K. M., and Shen, D. D. 1993. Contribution of probenecid-sensitive anion trasport processes at the brain caillary endothelium and choroid plexus to the efficient efflux on the valproic acid from the central nervous system. The Journal of Pharmacology and Experimental Therapeutics 268: 797-805.

Allendorf. F.W., P.R. England, G. Luikart, P.A. Ritchie, and N. Ryman. 2008. Genetic effects of harvest on wild animal populations. Trends in Ecology & Evolution 23: 327-337.

Baker, M. R. and D.E. Schindler. 2009. Unaccounted mortality in salmon fisheries: non-retention in gillnets and effects on estimates of spawners. Journal of Applied Ecology 46: 752-761.

Beamish, R., R. Sweeting, and and C-E. Neville . 2009. We Are on the Right Path, But It is Uphill Both Ways. Pages 45-62 in E.E. Knudsen and J.H. Michael Jr., editor. Pacific salmon environmental and life history models: advancing science for sustainable salmon in the future. American Fisheries Society, Symposium 71, Bethesda, Maryland.

Beverton, R. J. H. and and S.J. Holt. 1957. On the dynamics of exploited fish populations. United Kingdom Ministry of Agriculture and Fisheries Investigations, Ser. 2, Vol. 19.

Bilby, R. E. and others. Evaluating watershed response to land management and restoration actions: Intensively Monitored Waterhseds (IMW) Progress Report. 2004. The IMW Scientific Oversight Committee.

Clark, J. H., R. D. Mecum, A. McGregor, P. Krasnowski, and A.M. Carroll. 2006. The commercial salmon fishery in Alaska. Alaska Fishery Research Bulletin 12: 1-146.

Copes, P. 2000. Aboriginal fishing rights and salmon management in British Columbia; matching historical justice with the public interest. Pages 75-91 in E. E. Knudsen, C. R. Steward, D. D. MacDonald, J. E. Williams, and D. W. Reiser, editors. Sustainable fisheries management: Pacific salmon. Lewis, New York, New York.

Cowen, L., N. Trouton , and R. E. Bailey. 2007. Effects of Angling on Chinook Salmon for the Nicola River, British Columbia, 1996–2002 . North American Journal of Fisheries Management 27: 256-267.

Dann, T. H., C. Habicht, J.R. Jasper, H.A. Hoyt, A.W. Barclay, W.D. Templin, T.T. Baker, F.W. West, and L.F. Fair. 2009. Genetic Stock Composition of the Commercial Harvest of Sockeye Salmon in Bristol Bay, Alaska, 2006-2008. Alaska Department of Fish and Game, Fishery Manuscript Series No. 09-06, Anchorage, Alaska.

Dietrich, W. E. and F. Ligon. 2008. RIPPLE – A Digital Terrain-Based Model for Linking Salmon Population Dynamics to Channel Networks. Stillwater Sciences, Berkeley, California. http://www.stillwatersci.com/resources/RIPPLE_overview.pdf.

Farley, E. V. J., J.M. Murphy, M.D. Adkison, L. B. E. J. H. H., and J.H. Moss, a. J. N. 2007. Early marine growth in relation to marine-stage survival rates for Alaska sockeye salmon (Oncorhynchus nerka). Fisheries Bulletin 105: 121-130.

Greene, C. M. and and G.R. Pess. 2009. Multi-species modeling for salmon: alternatives, challenges, and opportunities. Pages 429-454 in E.E. Knudsen and J.H. Michael Jr., editor. Pacific salmon environmental and life history models: advancing science for sustainable salmon in the future. American Fisheries Society, Symposium 71, Bethesda, Maryland.

Gustafson, R. G., R.S. Waples, J.M. Myers, L.A. Weitkamp, G.J. Bryant, O.W. Johnson, and J.J. Hard. 2007. Pacific Salmon Extinctions: Quantifying Lost and Remaining Diversity. Conservation Biology 21: 1009-1020.

Hard, J. J., M.R. Gross, M. Heino, R. Hilborn, R.G. Kope, R. Law, and J.D. Reynolds. 2008. Evolutionary consequences of fishing and their implications for salmon. Evolutionary Applications 388-408.

Hilborn, R. 2009. Life history models for salmon management: the challenges. Pages 23-32 in E.E. Knudsen and J.H. Michael Jr., editor. Pacific salmon environmental and life history models: advancing science for sustainable salmon in the future. American Fisheries Society, Symposium 71, Bethesda, Maryland.

Hilborn, R. C. and C. J. Walters. 1992. Quantitative fisheries stock assessment, ed. Chapman and Hall. New York, New York.

Holt, K. R., S.P. Cox, and J. Sawada . 2009. In-Season Forecasting of Coho Salmon Marine Survival via Coded Wire Tag Recoveries. North American Journal of Fisheries Management 29: 1165-1182.

HSRG (Hatchery Scientific Reform Group). 2004. Hatchery Reform: Principles and Recommendations of the HSRG. Long Live the Kings, Seattle, Washington. http://www.lltk.org/pdf/hsrg/HSRG_Princ_Recs_Report_Full_Apr04.pdf.

Johnson, O. W. W. S. Grant R. G. Kope K. Neely F. W. Waknitz and R. S. Waples. 1997. Status Review of Chum Salmon from Washington, Oregon, and California. National Marine Fisheries Service, Northwest Fisheries Science Center, Conservation Biology Division, NOAA Technical Memorandum NMFS-NWFSC-32. http://www.nwfsc.noaa.gov/publications/techmemos/tm32/index.html.

Kaje, J., K. Rawson: A. Haas, and M. Crewson . 2008. Using the All-H Analyzer (AHA) Model to Advance H-Integration in the Snohomish Basin. Snohomish Basin Salmonid Recovery Technical Committee.

Knudsen, E. E. 2000. Managing Pacific salmon escapements: the gaps between theory and reality. Pages 237-272 in E. E. Knudsen, C. S. Steward, D. D. MacDonald, J. E. Williams,

and D. W. Reiser, editors. Sustainable fisheries management: Pacific salmon. Lewis, New York, New York.
Knudsen, E. E. and and E. G. Doyle. 2006. Science and technology are essential for sustaining salmon populations and their ecosystems. Pages 311-332 in Salmon 2100: The future of wild Pacific salmon. American Fisheries Society, Bethesda, Maryland, USA.
Knudsen, E. E. and J.H. Michael, Jr. 2009. Introduction: the past, present and future of Pacific salmon life history models. Pages 3-22 in E.E. Knudsen and J.H. Michael Jr., editor. Pacific salmon environmental and life history models: advancing science for sustainable salmon in the future. American Fisheries Society, Symposium 71, Bethesda, Maryland.
Kostow, K. 2009. Factors that contribute to the ecological risks of salmon and steelhead hatchery programs and some mitigating strategies. Reviews in Fish Biology and Fishereis 19: 9-31.
Lichatowich, J. A. and R.N. Williams. 2009. Failures to Incorporate Science into Fishery Management and Recovery Programs: Lessons from the Columbia River. Pages 1005-1019 in Pacific Salmon: Ecology and Management of Western Alaska's Populations . American Fisheries Society, Bethesda, Maryland.
McHugh, P., M. Baltzell, K. Kloempken, and L. Peterson . 2009. Marine Areas 5 and 6 Mark-Selective Recreational Chinook Fishery, Summer 2008 Post-season Report. Washington Department of Fish and Wildlife, Fish Program, REVISED DRAFT. February 17, 2009, Olympia, Washington 98501 . http://wdfw.wa.gov/fish/salmon/areas5_6msfchin2008_summer_revised_final_feb09.pdf.
Michael, J. H. Jr., A. Appleby, and J. Barr. 2009. Use of the AHA Model in Pacific salmon recover, hatchery, and fishery planning. Pages 455-464 in E.E. Knudsen and J.H. Michael Jr., editor. Pacific salmon environmental and life history models: advancing science for sustainable salmon in the future. American Fisheries Society, Symposium 71, Bethesda, Maryland.
Mobrand, L. E., Lichatowich, J. A., Lestelle, L. C., and Vogel, T. S. 1997. An approach to describing ecosystem performance "through the eyes of salmon". Canadian Journal of Fisheries and Aquatic Sciences 54: 2964-2973.
Newman, K. B. 2000. Hierarchic modeling of salmon harvest and migration. Journal of Agricultural and Biological Statistics: 430-435.
Newman, K. B. 1998. State-space modeling of animal movement and mortality with application to salmon. Biometrics 54: 1290-1314.
Noakes, D. J. and R.J. Beamish. 2009. Synchrony of Marine Fish Catches and Climate and Ocean Regime Shifts in the North Pacific Ocean. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 1: 155-168.
NRC (National Research Council). 1996. Upstream: salmon and society in the Pacific Northwest, ed. National Academy Press. Washington, District of Columbia.
Pauly, D. 1995.Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology and Evolution 10: 430.
PFMC (Pacific Fishery Management Council). 2007. Fisheries Regulation Assessment Model (FRAM) An Overview for Coho and Chinook. Pacific Fishery Management Council, V. 2, Portland, Oregon.
Policansky, D. and J.J. Magnuson. 1998. Genetics, metapopulations, and ecosystem management of fisheries. Ecological Applications, 8(1) Supplement, S119–S123.
PSIT and WDFW (Puget Sound Indian Tribes and the Washington Department of Fish and Wildlife). 2008. Chinook Harvest Management Performance Assessment: Management

years 2001-02 through 2006-07. Puget Sound Indian Tribes and the Washington Department of Fish and Wildlife.
PSIT and WDFW (Puget Sound Indian Tribes and the Washington Department of Fish and Wildlife). 2009. Comprehensive Management Plan for Puget Sound Chinook: Harvest Management Component .Puget Sound Indian Tribes and the Washington Department of Fish and Wildlife.
(PSSTRT) Puget Sound Steelhead Technical Review Team. 2005. Status review update for Puget Sound steelhead. Northwest Fisheries Science Center, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, Seattle, Washington. http://www.nwr.noaa.gov/Publications/Biological-Status-Reviews/upload/SR2005-steelhead.pdf.
Quinn, T. P., J.A. Peterson, V.F. Gallucci, W.K. Hershberger, and E.L. Brannon. 2002. Artificial Selection and Environmental Change: Countervailing Factors Affecting the Timing of Spawning by Coho and Chinook Salmon. Transactions of the American Fisheries Society 131: 591-598.
Ricker, W. E. 1975. Computation and interpretation of biological statistics of fish populations, ed. Fisheries Research Board of Canada Bulletin 191. Ottawa, Ontario.
Ricker, W. E. 1958. Maximum sustained yields from fluctuating environments and mixed stocks. Journal of the Fisheries Research Board of Canada 15: 991-1006.
Ruggerone, G. T., Farley, E., Nielsen, J. L., Ignell, S., and Hagen, P. 2005. Seasonal marine growth of Bristol Bay sockeye salmon in relation to competition with Asian pink salmon and the 1977 ocean regime shift. Fishery Bulletin 103: 355–370.
Ruggerone, G. T., Zimmerman, M., Myers, K. W., Nielsen, J. L., and Rogers, D. E. 2003. Competition between Asian pink salmon (Oncorhynchus gorbuscha) and Alaskan sockeye salmon (O. nerka) in the north Pacific Ocean. Fisheries and Oceanography 12: 209-219.
Scheuerell, M. D., R. Hilborn, M.H. Ruckelshaus, K.K. Bartz, K.M. Lagueux, A.D. Haas, and K. Rawson. 2006. The Shiraz model: a tool for incorporating anthropogenic effects and fish–habitat relationships in conservation planning. Canadian Journal of Fisheries and Aquatic Sciience 63: 1596–1607.
Smith, C. T., W.D. Templin, J.E. Seeb, and L.W. Seeb. 2005. Single Nucleotide Polymorphisms Provide Rapid and Accurate Estimates of the Proportions of U.S. and Canadian Chinook Salmon Caught in Yukon River Fisheries. North American Journal of Fisheries Management 25: 944–953.
SSDC (Shared Strategy Development Committee). 2007. Puget Sound Salmon Recovery Plan. Shared Strategy for Puget Sound, Plan adopted by the National Marine Fisheries Service (NMFS) January 19, 2007, Seattle, Washington. http://www.nwr.noaa.gov/Salmon-Recovery-Planning/Recovery-Domains/Puget-Sound/PS-Recovery-Plan.cfm.
Starr, P. and Hilborn, R. 1988. Reconstruction of harvest rates and stock contributions in the gauntlet salmon fisheries: Application to British Columbia and Washington sockeye (Oncorhynchus nerka). Canadian Journal of Fisheries and Aquatic Sciences 45: 2216-2229.
Van Doornik, D. M., D.J. Teel, D.R. Kuligowski, C.A. Morgan, and E. Casillas. 2007. Genetic Analyses Provide Insight into the Early Ocean Stock Distribution and Survival of Juvenile Coho Salmon off the Coasts of Washington and Oregon. North American Journal of Fisheries Management 27: 220-237.

Vander Haegen, G. E., C. E. Ashbrook, K. W. Yi, and J. F. Dixon .2004 . Survival of spring chinook salmon captured and released in a selective commercial fishery using gill nets and tangle nets . Fisheries Research 68: 123-133.
Wertheimer, A. 1988. Hooking mortality of chinook salmon released by commercial trollers. North American Journal of Fisheries Management 8: 346-355.
Zimmerman, M. Upl. Report. 2009 Wild Coho Forecasts for Puget Sound, Washington Coast, and Lower Columbia. Washington Department of Fish and Wildlife, Science Division, Olympia, Washington.

Evaluation of Protectin and Restoration Effectiveness

In this section we present a method for evaluating the effectiveness of the various protection and restoration strategies identified and described in the preceding chapters. This evaluation method is designed to be used to make recommendations and conclusions for implementing the most ecologically and fiscally effective strategies for restoring Puget Sound ecosystem function.

The goal of this suggested evaluation process is to evaluate how likely a particular strategy, or group of strategies, will achieve its stated goal; namely, the restoration or protection of one or more desirable attributes of Puget Sound. In short, how effective is the strategy in question?

The results of this evaluation should provide decision-makers with a clear indication of the relative effectiveness of different strategies; including those strategies which have been particularly effective and those where further improvement is needed. The evaluation will assist managers in deciding which strategies are thought to be most effective, their relative costs, extent or reliability of application, protection and restoration research needs, and guidance for monitoring.

1. Objectives

The objectives of the proposed evaluation process are to:

1. Develop an ecosystem and goal-based framework for classifying strategies to protect and restore Puget Sound (the framework should be consistent with the major topics identified in our outline);

2. Compile a list of strategies, organized by category (i.e., outline chapter) based on a review of the peer-reviewed literature. (Due to resource constraints, the compilation will not be exhaustive; rather, it is meant to enable us to broadly characterize strategies so that we can come up with a satisfactory method of organizing and evaluating them using the proposed assessment methodology.);

3. Identify performance criteria and develop a methodology (e.g., scoring procedures) for evaluating the effectiveness of individual or groups (combinations) of strategies to protect and restore Puget Sound. The approach should be rational, adaptable, easily comprehended, and capable of being applied at different scales (e.g., it can be used by PSP and local governments to assess the effectiveness of their restoration and protection strategies);

4. Apply the criteria and scoring procedures to obtain a qualitative ranking for each strategy or group of strategies;

5. Provide a foundation for effectiveness monitoring and adaptive management; the framework should be based on recommended indicators and existing governance systems (e.g., State, PSP, WRIAs) (This is beyond the scope of our current assignment, but the method and results of our evaluation should inform and integrate with future monitoring and management.); and

6. Provide practical guidance so that others can evaluate the effectiveness of restoration and protection strategies without direct assistance from us. This, too, is outside our current scope of work. However, the evaluation methodology is intended to be reapplied iteratively in the future as new goals are formulated and new information is developed.

Proposed Methodology

The categorization of strategies and development of appropriate assessment criteria is proposed to proceed in two steps. Step 1 is the articulation of the basic goals of restoration and protection and Step 2 is the development of performance criteria that help us define and evaluate effectiveness, the main focus of the process described here.

Comparison to Performance Goals

We propose that the evaluation be conducted relative to three broad sets of pre-stated goals for protection and restoration:

- Perceived technical performance and scientific soundness
- As compared to the PSP Action Agenda Priorities
- Relative to the PSP Action Agenda Outcomes

Categorization of Strategies

Because the interaction of protection and restoration strategies with the actual implementation, as reflected in habitat outcomes, are extremely complex, dynamic, and at variable scales, a method is needed to account for as many features as possible in the evaluation. We therefore propose that each strategy that is evaluated be first categorized according to area, method, and scale of application, as follows:

Target area of application

- Watersheds and tributaries
- Estuarine and marine
- Fish and wildlife populations
- Overarching or general

Method of application

- Preservation
- Protective retrofit actions
- Protective new development and redevelopment actions
- Physical habitat restoration
- Policy changes
- Public Education

Scale of application

- Overarching - Applies broadly to all areas (e.g., a broad policy change)
- Regional (e.g., several watersheds or large portions of Puget Sound)
- Local (e.g., one watershed or tributary, or Puget Sound bay)
- Site or population-specific

Evaluation Criteria

The results of such an evaluation process would include tabular matrices that list the known protection and restoration strategies and rate them under each of the preceding categories. In this way, managers will be better informed about which strategies are thought to be most effective, their breadth or reliability of application, and protection and restoration research needs.

The proposed assessment approach for each strategy will consist of a set of criteria which, taken together, provide the basis for an assessment of the particular strategy under consideration. To do this, a summary assessment would be conducted and each strategy rated for its status according to the following rating criteria (expressed positively so that all metrics will have the same sign or direction):

1. Perceived relative effectiveness
2. Level of scientific basis (alternative: research needs)
3. Certainty of success (alternative: risk)
4. Confidence in outcomes (alternative: uncertainty)
5. Low need for monitoring
6. Degree to which currently monitored
7. Low total cost
8. Benefits in relation to costs
9. Consistent with existing processes
10. Extent of existing application, i.e., level of participation, commitment, ownership, compliance, etc. and
11. Application to multiple threats
12. Capacity – technical and financial resources, etc.
13. Informs monitoring, learning, and adaptive management

Some of these criteria overlap with others; they should be refined to a handful of non-redundant, easily intuited criteria that are applicable to all strategies. The final set of criteria should allow us to objectively assess the performance and outcomes of the strategies to which they relate. And finally, they should be comprehensible to other, less technically oriented individuals and stakeholders.

Rating of Strategies

The evaluation system requires scoring metrics and a process by which individual evaluators are able to review available information and indicate the extent to which each criterion is (or is likely to be) met. It would help if the criteria were framed as a series of questions that ensured that all aspects and dimensions of the strategy are considered.

We propose the following scoring metrics:

4 = the criterion is fully met

3 = the criterion is mostly met, but further improvements can be achieved

2 = the criterion has only partially been met, there is potential for further improvements

1 = the criterion has been barely met; but there is promise for the future

0 = the criterion has not been met; further improvements are unlikely

In designing the final evaluation, we propose the strategies would be listed in the matrices presented in tabular form as illustrated below, which combine all the features described above. In these tables, we rate each strategy in terms of its subjectively determined effectiveness.

Final Evaluation

The final evaluation of a given strategy combines consideration for the type and extent of the strategy with its rated performance as a scientifically substantiated strategy to gather with its perceived satisfaction of Action Agenda outcomes and priorities. We also recognize that the system posed here has certain drawbacks. For example, the rating score given to any one strategy for a certain category is highly dependent on the setting where the rated strategy would be applied and ecological and economic details of the particular application. Therefore it may be necessary to refine or revise the rating system to include additional considerations for how, where and by whom the application would be implemented. For now, however, this suggested process may be useful as a starting point for evaluation of protection and restoration strategies.

Updating protection and restoration performance over time

There would also be a need for periodically updating this information so it can be used to inform management decisions over time; i.e., maximizing the effectiveness of an integrated research, monitoring, and adaptive management program. The approach described above provides the Partnership with a vehicle for future evaluation and management of protection and restoration strategies into the future since it can be continually updated and revised.

Appendices

1. Appendix 4A: Elements of watershed-based strategies, links to PSP results chains (Neuman et al. 2009)

Box A1. Major Elements of a Watershed-Based Strategy

- A watershed instead of political-boundary basis.
- Centralizing responsibility and authority for implementation with a municipal lead permittee working in partnership with other municipalities in the watershed as co-permittees—RC6 (Stormwater) C2, specifically C2(2) (inform and support implementation and adoption of NPDES permits).
- Embracing the full range of sources of aquatic ecosystem problems now usually under uncoordinated management and permitting; integration of all local water permits under the co-permittee system organized by watersheds—RC6 (Stormwater) C2, specifically C2(9) (implement NPDES industrial permits, WSDOT permits, DOE oversight).
- Extending full permit coverage, as appropriate, to any area in the watershed zoned or otherwise projected for development at an urban scale (e.g., more than one dwelling per acre)—RC6 (Stormwater) C2.
- Comprehensively covering all stages of urbanization: construction, new development, redevelopment, retrofit—RC1 (Land Protection) A2, specifically A2.2.8 (develop incentives to increase and improve redevelopment within UGSs); RC6 (Stormwater) C2, specifically C2(6) (retrofit stormwater systems).
- Adopting a minimum goal in every watershed to avoid any further loss or degradation of designated beneficial uses within the watershed's component water bodies.
- Assessing water bodies that are not providing designated beneficial uses in order to set goals aimed at recovering these uses—RC1 (Land Protection) A1, specifically A1(3) (initiate and complete watershed assessments); RC2 (Flow Protection) A3.
- Defining careful, complete, and clear beneficial-use-attainment objectives to be achieved as the essential compliance endpoints.
- Concern with water quantity along with water quality—RC2 (Flow Protection) A3;
- Efficient, advanced scientific and technical watershed analysis to identify negative impact sources and set objectives and strategies—RC1 (Land Protection) A1, specifically A1(3) (initiate and complete watershed assessments); RC2 (Flow Protection) A3.
- Strategies to emphasize maximum isolation of receiving waters from impact sources; i.e. maximize application of low-impact development (LID) (retitled by the committee Aquatic Resources Conservation Design, ARCD) principles and methods—RC2 (Flow Protection) A3, specifically A3.3.2 (allow and promote rainwater harvesting) and A3 new strategies; RC6 (Stormwater) C2, specifically C2(3) (assist cities and counties in incorporating LID into all stormwater codes), C2(4) (develop and implement LID incentives), C2(6) (retrofit stormwater systems), and C2(8) (private stewardship and incentives for pollution prevention).
- Assigning municipalities more responsibility, along with more authority and funding, for the range of sources within their jurisdictions.

- Developing and appropriate allocating funding sources to enable municipalities to implement effectively—RC1 (Land Protection) A2, specifically A2(5) and A2(8) (both funding and technical assistance).
- A monitoring system composed of direct measures to assess compliance and progress toward achieving objectives and diagnosing reasons for the ability or failure to meet objectives, along with a research component to address information gaps—RC6 (Stormwater) C2, specifically C2(1) (establish regional coordinated monitoring program for stormwater under NPDES).
- Organizing consortia of agencies to design and conduct monitoring programs—RC6
- (Stormwater) C2, specifically C2(1) (establish regional coordinated monitoring program for stormwater under NPDES).
- An adaptive management framework to apply monitoring results and make early course corrections toward meeting goals and objectives, if necessary.
- A system of in lieu fees and trading credits to compensate for legitimate inability to meet requirements on-site by supporting equivalent effort elsewhere within the same watershed.

In addition to the Results Chain strategies denoted in the list, the NRC committee's recommended program could serve as a framework to promote strategies RC1 (Land Protection) A1, specifically A1(1) (convene regional planning forum for coordinated vision), and RC2 (Flow Protection) A3, specifically A3.2 (reform state water laws). Implementation of other Results Chain strategies probably could also benefit, although perhaps less directly, from the recommendations in the NRC (2009) report.

Appendix 4B: Recommendations from Booth et al. (2001) and Horner, May and Livingston (2003)

Horner, May and Livingston (2003) put forward the following recommendations based on their data and the trends signified within them:

1. Systematically collect data on regionally representative stream benthic macroinvertebrate and fish communities. Extend the program's coverage over the full range of urbanization. Use the data to develop regionally appropriate biological community indices.

2. Develop a geographic information system to organize and analyze watershed land use and land cover (LULC) data. Collect data on regionally appropriate LULC variables, particularly measures of impervious and forested cover in the watershed as a whole, at least two riparian bands extending to points relatively near and far from the stream, and in other local areas fairly close to the stream.

3. Base stream watershed management on specific objectives tied to desired biological outcomes.

4. If the objective is to retain an existing levels of stream function, very broadly preserve the extensive watershed and riparian natural vegetation and soil cover almost certainly present through mechanisms like outright purchase, conservation easements, transfer of development rights, etc.

5. If the objective is to prevent further degradation when partially developed areas urbanize more, maximize protection of existing natural vegetation and soil cover in areas closest to the stream, especially in the nearest riparian band. In the uplands, generally develop in locations already missing characteristic natural vegetation. As much as possible, preserve existing natural cover and limit conversion to impervious surfaces. The lower the level of existing development, the more important it is to protect existing natural vegetation and soil cover

6. In addition, fully serve newly developing and redeveloping areas with stormwater quantity and quality control best management practices (BMPs) sited, designed, and operated at state-of-the-art levels. Attempt to retrofit these BMPs in existing developments. The higher the level of existing development, the more important it is to control stormwater, since extensive land conversion results in the loss of natural vegetation and soil cover..

7. Where riparian areas have been degraded by encroachment, crossings, or loss of mature, natural vegetation, give high priority to restoring them to extensive, unbroken, well vegetated zones. This strategy could be the most effective, as well as the easiest, step toward improving degraded stream habitat and biology. Riparian areas are more likely to be free of structures than upland areas and more directly influence stream ecology. Also, riparian restoration fits well with other objectives, like flood protection and provision of wildlife corridors and open space.

Recommendations from Booth et al. 2001

Booth et al. (2001) interpreted their results to devise explicit strategies for protecting and restoring Puget Sound's tributary streams, starting with a set of general strategies applying over the gradient of urbanization:

1. Recognize and preserve high-quality, low-development watershed areas.

2. Aggressively (and completely) rehabilitate streams where recovery of ecosystem elements and processes is possible. This condition is likely to be met only in low-development areas that happen to have relatively low to moderate levels of ecological health, because the agents of degradation are probably easier to identify and more amenable to correction.

3. Rehabilitate selected elements of mid-range urban watersheds, where complete recovery is not feasible but where well-selected efforts may yield direct improvement, particularly in areas of public ownership.

4. Improve the most degraded streams by first analyzing the acute cause(s) of degradation, but recognize that the restoration potential for populations of original in-stream biota is minimal.

5. In the most highly developed watersheds, education and/or community outreach is not just appropriate but crucial. Here, the level of public interest is likely to be highest, stream-side residents have greater direct individual influence over whether healthy stream conditions are maintained, and most of the riparian corridor is not under public ownership or control.

Booth et al. (2001) went on to offer specific recommendations for rehabilitation efforts:

1. Make direct, systematic, and comprehensive evaluation of stream conditions in areas of low to moderate development.

2. Recognize that the hydrologic consequences of urban development cannot be reversed without extensive redevelopment of urban areas. Likewise, the recovery of physical and biological conditions of streams is infeasible without hydrologic restoration over a large fraction of the watershed land area. This conflict can be resolved only if there are particular, ecologically relevant characteristics of stream flow patterns that can be managed in urban areas. Effective hydrologic mitigation will require approaches that can: (1) delay the timing of storm-flow discharges in relatively small storms, and (2) store significant volumes of rain for at least days or weeks. In the long run the goal should be to mimic the hydrologic responses across the hydrograph and not just truncate the high or low flow components. The rate of rise and decline of the hydrograph is just as important as the existence of peaks and lows. This approach almost certainly requires greater reliance on on-site storage to better emulate the hydrologic regime of undisturbed watersheds, either through dispersed infiltration, on-site detention, or forest preservation.

3. Where overall basin development is low to moderate, natural riparian corridors have significant potential to maintain or improve biological condition. Protecting high quality wetland and riparian areas that persist in less developed basins may also serve as a source of colonists (e.g., plants, invertebrates, fish) to other local streams that are subject to informed restoration efforts. At the same time, even small patches of urban land conversion in riparian areas can severely degrade local stream biology. As both a conservation and restoration strategy, protection and revegetation of riparian areas is critical for preventing severe stream degradation, but these measures alone are not adequate to maintain ecosystem function in streams draining highly urban basins.

Synthesis of Stream Watershed Management Strategies

Table B1 presents, in four categories, the elements of strategies drawn from the recommendations developed from the two large research projects on Puget Sound watersheds and streams (see above for fuller descriptions). It gives general notes regarding estimating the probable effectiveness and relative certainties associated with major strategies and references to sources of more information. Table B1 also relates the various strategies given here with those in the Results Chain memo. The strategies in Table B1 address multiple threats to the Puget Sound ecosystem, including stream channel hydromodification, salmon spawning and rearing habitat degradation, stream food web disruption, acute and chronic toxicity effects on aquatic organisms from metal and organic pollutants and increased pollutant loadings to all downstream waters, including Puget Sound.

Table B1. Strategies for Watershed Management to Protect and Restore Puget Sound's Stream Tributaries

Table B1. Strategies for Watershed Management to Protect and Restore Puget Sound's Stream Tributaries

Category	Strategies	Notes	Associated Results Chain Strategies
Database development	• Stream biological communities • Land use and cover		RC1-A1(3); RC4-A1(3)
Establish objectives for an integrated approach	• For streams with high biological integrity • For streams with reduced biological integrity	• Appropriate objectives would be to retain existing WCI and/or forest cover and EIA balance. • Appropriate objectives would be to retain existing WCI or recover a selected WCI and/or forest cover and EIA balance.	RC6-C2
Manage watersheds of streams with high biological integrity	Preserve existing watershed and riparian vegetation and soil cover through land use purchase, planning, and regulatory mechanisms.	Estimate effectiveness and relative certainty according to the data and methods presented by Horner, May, and Livingston (2003) and Booth, Harley, and Jackson (2002).	RC1-A1, A2, A3, A4, A4(6)
Manage watersheds of streams with reduced biological integrity	• Maximize protection of existing vegetation and soil closest to the stream. • Restore riparian areas to extensive, unbroken, well vegetated zones. • Emphasize development in already disturbed locations. • Serve newly developing and redeveloping areas with state-of-the-art stormwater quantity and quality control BMPs, especially low-impact development types. • Retrofit existing development with these BMPs. • Perform in-stream rehabilitation as appropriate to watershed conditions. • Conduct watershed resident education.	• Estimate effectiveness and relative certainty according to the data and methods presented by Horner, May, and Livingston (2003) and Booth, Harley, and Jackson (2002). • Refer to stormwater management segment of this chapter below for information on effectiveness and relative certainty. • Refer to stream restoration segment of this chapter below for information on effectiveness and relative certainty.	RC4-B1, B1(1), B1(3) RC2-A3.3.2, A3 new strategies; RC6-C2(3), C2(4), C2(6), C2(8) RC4-B1(3)

Appendix 4C: Supporting material for effectiveness and relative certainty of wetland management efforts

Water level fluctuation (WLF) was computed as the difference between crest stage and average base stage. Crest stage was determined with a crest-stage gauge, which records the maximum stage in a time period through the deposition level of cork dust on a plastic tube within a pipe housing. Average base stage was calculated as the mean of the stage at the beginning and end of the time period. WLF statistics were compute over extended time intervals involving a number of separate determinations. Table C1 depicts the relationship calculated by Chin (1996) and (Horner et al. 2001) between mean annual WLF and watershed TIA. Clearly, the two variables are not independent, as installation of impervious cover often accompanies removal of forest. Loss of watershed forest cover has been shown to be an important factor driving increases in WLF (Reinelt and Taylor 2001).

Table C1. Relationship Between Mean Annual Water Level Fluctuation (WLF) and Watershed Total Impervious Area (TIA) (after Chin 1996, Horner et al. 2001)

Mean Annual WLF Was:	If TIA Was:	Cases Where True:
< 20 cm	< 6%	100%
> 20 cm	> 21%	89%
> 30 cm	> 21%	50%
> 30 cm	> 40%	75%
> 50 cm	> 40%	50%

Appendix 4D: Supporting material for lake management strategies

Box D1. Algal biomass control techniques from Cook et al. (2005).

- Nutrient diversion (removal or treatment of direct external inputs);
- Protection from diffuse nutrient sources (e.g., urban, agricultural, and forestry stormwater runoff);
- Dilution (to reduce nutrient concentrations) and flushing (to increase water exchange rate and consequent algal cell washout);
- Hypolimnetic (lower thermal layer) withdrawal (to discharge nutrient-rich water resulting from sediment release in the low-oxygen environment of thermal stratification);
- Phosphorus inactivation (precipitation by aluminum salt addition) and sediment oxidation (calcium nitrate injection to stimulate denitrification and oxidize organic matter);
- Biomanipulation (managing other trophic levels [zooplankton, fish] to control algae); and
- Copper sulfate (algicide) addition.

Macrophyte control mechanisms covered by Cooke et al. (2005) are:

- Restoring desirable plants to replace undesirable ones;
- Water level drawdown (to desiccate undesirables);

- Preventing invasion and physically removing undesirables;
- Sediment covers and surface shading
- Chemical controls; and
- Biological controls (insects, fish, other).

Three methods convey multiple benefits:

- Hypolimnetic aeration and oxygenation (to raise oxygen content and open habitat to cold-water fish; also to reduce sediment phosphorus release);
- Artificial circulation (use pumps, jets, or diffused air for the same purposes, plus move algal cells out of the lighted zone); and
- Sediment removal (for deepening, nutrient control, toxic substances removal, and/or rooted macrophyte control).

Appendix 4E: Supporting information on ARCD strategies

Stages of urbanization and their effects on ARCD strategies

From the NRC report (2009, p405-406):

In water bodies that are not in attainment of designated uses, it is likely that the physical stresses and pollutants responsible for the loss of beneficial uses will have to be decreased, especially as human occupancy of watersheds increases. Reducing stresses, in turn, entails mitigative management actions at every life stage of urban development: (1) during construction when disturbing soils and introducing other contaminants associated with building; (2) after new developments on Greenfields are established and through all the years of their existence; (3) when any already developed property is redeveloped; and (4) through retrofitting static existing development. Most management heretofore has concentrated on the first two of those life stages.

The proposed approach recognizes three broad stages of urban development requiring different strategies: *new development*, *redevelopment*, and *existing development*. New development means building on land either never before covered with human structures or in prior agricultural or silvicultural use relatively lightly developed with structures and pavements (i.e., Greenfields development). Redevelopment refers to fully or partially rebuilding on a site already in urban land use; there are significant opportunities for bringing protective measures to these areas where none previously existed. The term existing development means built urban land not changing through redevelopment; retrofitting these areas will require that permittees operate creatively. What is meant by redevelopment requires some elaboration. Regulations already in force typically provide some threshold above which stormwater management requirements are specified for the redeveloped site.

All urban areas are redeveloped at some rate, generally slowly (e.g., roughly one or at most a few percent per annum) but still providing an opportunity to ameliorate aquatic resource problems over time. Extending stormwater requirements to redeveloping property also gradually “levels the playing field” with new developments subject to the requirements. … Some jurisdictions offer exemptions from stormwater management requirements to stimulate desired economic activities or realize social benefits. Such exemptions should be considered very carefully with respect to firm criteria designed to weigh the relative socioeconomic and environmental benefits, to prevent abuses, to gauge just how instrumental the exemption is to gaining the socioeconomic benefits, and to compensate through a trading mechanism as necessary to achieve set aquatic resource objectives.

It is important to mention that not only residential and commercial properties are redeveloped, but also streets and highways are periodically rebuilt. Highways have been documented to have stormwater runoff higher than other urban land uses in the concentrations and mass loadings of solids, metals, and some forms of nutrients (Burton and Pitt, 2002; Pitt et al., 2004; Shaver et al., 2007). Redevelopment of transportation corridors must be taken as an opportunity to install storm-water control measures (SCM) effective in reducing these pollutants.

Opportunities to apply SCMs are obviously greatest at the new development stage, somewhat less but still present in redevelopment, but most limited when land use is not changing (i.e., existing development). Still, it is extremely important to utilize all readily available opportunities and develop others in static urban areas, because compromised beneficial uses are function of the development in place, not what has yet to occur. Often, possibly even most of the time, to meet watershed objectives it will be necessary to retrofit a substantial amount of the existing development with SCMs. To further progress in this overlooked but crucial area, the Center for Watershed Protection issued a practical Urban Stormwater Retrofit Practices manual (Schueler et al., 2007).

Application of ARCD for Construction and Industrial Land Uses

From the NRC (2009) report:

All of the principles discussed above apply to industrial and construction sites as well: minimize the quantity of surface runoff and pollutants generated in the first place, or act to minimize what is exported off the site. Unfortunately, construction site stormwater now is managed all too often using sediment barriers (e.g., silt fences and gravel bags) and sedimentation ponds, none of which are very effective in preventing sediment transport. Much better procedures would involve improved construction site planning and management, backed up by effective erosion controls, preventing soil loss in the first place, which might be thought of as ARCD for the construction phase of development. Just as ARCD for the finished site would seek to avoid discharge volume and pollutant mass loading increase above pre-development levels, the goal of improved construction would be to avoid or severely limit the release of eroded sediments and other pollutants from the construction site.

Other industrial sites are faced with some additional challenges. First, industrial sites usually have less landscaping potentially available for land-based treatments. Their discharges are often more contaminated and carry greater risk to groundwater. On the other hand, industrial operations are amenable to a variety of source control options that can completely break the contact between pollutants and rainfall and runoff. Moving operations indoors or roofing outdoor material handling and processing areas can transform a high-risk situation to a no-risk one. It is recommended that industrial permits strongly emphasize source control (e.g., pollution prevention) as the first priority and the remaining ARCD measures as secondary options. Together these measures would attempt to avoid, or minimize to the extent possible, any discharge of stormwater that has contacted industrial sources.

It is likely that the remaining discharges that emanate from an industrial site will often require treatment and, if relatively highly contaminated, very efficient treatment to meet watershed objectives. Some industrial stormwater runoff carries pollutant concentrations that are orders of magnitude higher than now prevailing water quality standards. In these cases meeting watershed objectives may require providing active treatment, which refers to applying specifically engineered physicochemical mechanisms to reduce pollutant concentrations to reliably low levels (as opposed to the passive forms of treatment usually given stormwater, such as ponds, biofiltration, and sand filters). Examples now in the early stages of application to stormwater include chemical coagulation and precipitation, ion exchange, electrocoagulation, and filtration enhanced in various ways. These practices are undeniably more expensive than source controls and other ARCD options and traditional passive treatments. If they must be used at all, it is to the advantage of all parties that costs be lowered by decreasing contaminated waste stream throughput rates to the absolute minimum.

Appendix 4F: Supporting information on international Best Management Practices (BMP)

Appendix 4F. Supporting information on international Best Management Practices (BMP)

Table F1. Statistics on Conventional Stormwater BMP Effluent Water Quality from the International Stormwater Best Management Practices Database[a].

Pollutant[b]	Detention Ponds[c]	Wet Ponds	Treatment Wetlands	Biofilters[c]	Media Filters[c]	Hydrodyn. Devicesc
TSS	31 (16-46)	13 (7-19)	18 (9-26)	24 (15-33)	16 (10-22)	38 (21-54)
T N	2.72 (1.81-3.63)	1.43 (1.17-1.68)	1.15 (0.82-1.62)	0.78 (0.53-1.03)	0.76 (0.62-0.89)	2.01 (1.37-2.65)
T P	190 (120-270)	120 (90-160)	140 (40-240)	340 (260-410)	140 (110-160)	260 (120-480)
D P	120 (70-180)	80 (40-110)	170 (30-310)	440 (210-670)	90 (70-110)	90 (40-130)
T Cu	12.1 (5.4-18.8)	6.4 (4.7-8.0)	4.2 (0.6-7.8)	10.7 (7.7-13.7)	10.2 (8.2-12.3)	14.2 (8.3-20.0)
T Zn	60 (21-100)	29 (21-38)	31 (13-67)	40 (28-52)	38 (17-58)	80 (53-107)
T Pb	15.8 (4.7-26.9)	5.3 (1.6-9.0)	3.3 (2.3-4.2)	6.7 (2.8-10.6)	3.8 (1.1-6.4)	10.6 (4.3-16.9)
D Cu	7.4 (3.3-11.5)	4.3 (3.7-5.7)	No data	8.4 (5.7-11.5)	9.0 (7.3-10.7)	13.9 (4.4-23.4)
D Zn	26 (11-41)	33 (18-48)	No data	25 (19-32)	51 (29-73)	42 (10-75)
D Pb	2.1 (0.9-3.2)	2.5 (1.6-9.0)	0.9 (0.85-0.89)	2.0 (1.3-2.7)	1.2 (0.8-1.6)	3.3 (2.2-4.5)

[a] Median (95% confidence limits), with units in μg/L, except for TSS and Total N (mg/L); "no data" indicates insufficient reports to compute statistics.

[b] TSS—total suspended solids, T—total, N—nitrogen, P—phosphorus, D—dissolved, Cu—copper, Zn—zinc, Pb—lead, Cd—cadmium

[c] Detention ponds have a range of residence times from hours to 3 days; biofilters represent a range of vegetated conveyance configurations; media filters generally have sand as the medium; hydrodynamic. devices—hydrodynamic devices of various designs.

Appendix 4G: Supporting information on removal of fecal coliforms from stormwater runoff

REMOVAL OF FECAL COLIFORMS FROM STORMWATER RUNOFF:

A LITERATURE REVIEW

Report to City of Blaine

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INTRODUCTION

SCOPE OF REVIEW

Urban stormwater runoff is a widely recognized source of shellfish contamination by potential disease-causing organisms, which can lead to the closure of beds to harvest for human consumption. The literature search was intended to provide a current portrait of management options to reduce shellfish bed pathogen contamination problems associated with urban stormwater. More specifically, it concentrated on stormwater treatment methods that could be investigated further by the City of Blaine, Washington to protect shellfish harvest areas in adjacent marine waters.

The review encompassed exploring scientific and technical research databases provided by University of Washington Libraries, as well as the Internet using Google. Research databases accessed included Environmental Engineering Abstracts, Water Resources Abstracts, National Technical Information Service (NTIS), and U.S. Environmental Protection Agency (EPA) Publications Online.

The principal keyword used in the search was “fecal coliforms”, because of the prominence of this contamination indicator group in assessing shellfish bed status. The investigation did not use the broader categories “bacteria” and “pathogens” or specific microorganisms; but items reported in these terms were collected if they appeared to be relevant. “Stormwater”, “treatment”, “removal”, and “reduction” were used as secondary delimiters when necessary to narrow the inquiry to items of most direct interest.

BACKGROUND

Fecal coliforms represent a group of bacteria that have long been used as indicators of contamination by a whole host of potentially disease-causing microorganisms. Their popularity is mainly because: (1) they are relatively easy and inexpensive to measure; and (2) they have an association, and sometimes a demonstrated statistical correlation, with pathogenic organisms (Kadlec and Knight 1996).

The use of fecal coliforms (FCs) to indicate possible disease agents is not a perfect solution for several reasons. They originate from the intestinal tracts of all warm-blooded animals, and thus do not necessarily indicate human disease potential. Virulent pathogens, especially viruses, can be present even with relatively low FCs or absent with comparatively high values. Furthermore, FCs are very dynamic and responsive to a number of variables in the natural environment, such as temperature, growth substrate, and the kinetic energy of flow or currents. Nevertheless, no feasible replacement for routine monitoring has emerged, and FCs are the most common basis for regulating and managing aquatic resources. Because of this standard, FCs were taken as the basis for this literature review.

FCs fit within the broader group termed total coliforms, some of which have sources other than animal intestines (e.g., natural soils). The *Escherichia coli*, a subset of the fecal coliforms, are sometimes used as an alternative indicator, especially outside the United States. Other bacterial groups that have served this purpose include the enterococci and fecal streptococci. This review concentrates on FCs because of their general prominence as an index of pathogen contamination and their specific importance in water quality management within Drayton Harbor and the City of Blaine itself. In addition, the City of Blaine has acquired historical FC data within Drayton Harbor that can be used for monitoring the effectiveness of stormwater best management practices (BMPs) that are developed in the future. Data on other indicators are reported when they appear in the references consulted for information on FCs.

Evidencing their variability, FCs in urban stormwater runoff can range over a number of orders of magnitude. They most commonly fall into a range of about 10^2-10^4 colonies/100 mL (henceforth to be abbreviated as n/100 mL). However, values of <10 and as high as ~10^6/100 mL are not uncommon (Schueler 1999a). Relatively high values are usually associated with a sewage release through an event like septic system failure, sanitary sewer overflow, or illicit connection (Pitt 1998). The mean in wide-ranging large data sets has been reported as approximately 15,000-20,000/100 mL (Pitt 1998, Schueler 1999a).

To protect shellfish harvesting in the State of Washington, Chapter 173-201A WAC requires that the geometric mean of FC readings in shellfish waters not exceed 14/100 mL, with no more than 10 percent of the measurements surpassing 43/100 mL. It is clear that to meet these criteria, typical concentrations in urban stormwater must be greatly decreased in almost any case, perhaps only excepting the most expansive and well flushed receiving waters. Reduction of mean concentration by 99 percent would still leave FCs at 150-200/100 mL, an order of magnitude higher than a 14/100 mL target. Therefore, FCs in typical urban stormwater must be reduced by source control, treatment, or both to levels more like 99.9 percent to assure protection of shellfish resources.

FCs generally fall in the range 10^5-10^7/100 mL in municipal wastewater effluents following both primary and secondary treatments but before disinfection. The distinction in concentrations between stormwater and wastewater is important, because the efficiency of reduction (percent removal) in a treatment system depends in part on the influent concentration; i.e., a higher efficiency in percentage terms is frequently registered with a "dirtier" than a "cleaner" influent. This phenomenon has been widely observed in stormwater treatment systems for various contaminants. What is also often seen is that the ultimate effluent quality produced by a treatment is comparable with varying influent concentrations and efficiencies. Therefore, effectiveness of a stormwater best management practice should be gauged in terms of both efficiency and consistently produced effluent quality.

Information is more scarce for FCs and other bacteriological measures than for other common stormwater contaminants, both for initial and treated runoff quality. This scarcity is due primarily to the relatively short holding time before microbiological analyses must start (6 hours) and the need to disinfect any surfaces that a sample contacts during monitoring. This period is shorter than typical full storm lengths, especially in the Pacific Northwest (the mean wet season length is 21 hours in Seattle). It would be very difficult to disinfect all of the tubing and surfaces in automatic sampling equipment. It is therefore virtually impossible to generate full storm composite samples for FC analysis. Monitoring must rely on a single grab sample, which is an unlikely representative of the overall event, or a series of burdensome grab samples taken throughout the storm and composited in relation to simultaneous flow measurements. The relative variability of FCs, and their consequent high statistical variance, also impedes obtaining data from which decisive conclusions can be drawn.

Broadly speaking, the bacterial content of stormwater runoff can be restricted by source controls, treatment BMPs, or both. Source controls are means of preventing contact between contaminants and rainfall or runoff. Hence, they are preventive practices; if there is complete lack of contact, they are 100 percent effective. Treatment BMPs are engineered devices intended to remove pollutants after they have already entered runoff. The principal types are constructed wetlands, ponds of various configurations, swales or surfaces that expose pollutants to vegetation and soil where pollutant removal mechanisms operate, and media filters. It is impossible according to inviolate physical laws to recapture all substances once released. Therefore, treatment BMPs having a surface discharge are never 100 percent effective in preventing delivery of pollutants to the receiving water.

The most common BMP investigated for bacteria reduction is some form of constructed wetland, with ponds being second in frequency. Both of these treatment systems have extended residence times, generally some days in length. The entering and exiting water streams are thus from different storms. Nevertheless, many studies compare influent and effluent quality without accounting for this fact. This failing is particularly evident in bacteria sampling because of the near impossibility of compositing samples from different points in time.

This literature review considers these data collection issues and interprets the utility of the results accordingly. Caution is applied when reporting results gained through incomplete sampling or from theoretical considerations with no or insufficient empirical demonstration.

EFFECTIVENESS OF STORMWATER BMPS IN FECAL COLIFORM REDUCTION

PRE-2000 EXPERIENCE

Schueler (1999b) summarized the experience in treating stormwater for FC reduction through the late 1990s. He covered sources, removal mechanisms, BMP treatment abilities, and recommendations for improving the quality of discharges to receiving waters from the pathogen standpoint. This review draws mostly from the last two topics. Schueler's summary was based on 24 performance studies representing 10 stormwater ponds, nine sand filters, and five biofiltration swales. Most, but not all, focused on fecal coliforms, and grab sampling was the usual monitoring technique.

In Schueler's database mean pond efficiency for FCs was 65 percent (range –5 to 98 percent). The corresponding figures for sand filters were mean 50 percent and a range of –68 to 97 percent). Swales generally discharged higher FC concentrations than entered (mean removal –58 percent). Pet wastes and *in situ* multiplication of bacteria were cited as the primary reason for poor swale performance. Schueler also reported effluent concentrations, with the means being 5144/100 mL for ponds, 5899/100 mL for sand filters, and 2506/100 mL for swales. It is apparent that influent concentrations were generally lower in the few swale studies than in the more numerous accounts for the other two BMPs.

The results indicate that ponds and sand filters can reduce stormwater bacterial contamination but not in a consistent and reliable manner. Effluent concentrations were still of the order 10^3/100 mL, much higher than shellfish water quality criteria (~10^1/100 mL). It is true that dilution could lower the concentration sufficiently to meet criteria, but on the other hand it is also true that continuing large inputs of viable organisms would form a basis for sustaining a reproducing bacterial community in the receiving water.

Schueler concluded with a number of recommendations to improve performance. They included BMP structural modifications but highlighted source controls as means to prevent contaminant introduction in the first place. It would appear to be unlikely that effluents could be improved to the ~10^1/100 mL levels with structural fixes of these conventional BMPs alone. If this level is to be reached, some combination of highly effective source controls and advanced treatment BMPs will be needed.

The experience in using constructed wetlands to treat domestic wastewater can offer some insights applicable to stormwater. Kadlec and Knight (1996) covered all aspects of that topic following an intensive period of research on the subject. They summarized 21 studies in which fecal coliforms were measured before any disinfection. Reduction efficiencies ranged from <0 to 99.9 percent, 67 percent above 95 percent. However, the great majority of effluent concentrations were still of the order 10^2/100 mL, including all but one case in the group having efficiency exceeding 95 percent. The authors concluded that outflow concentrations cannot be reduced to near zero without disinfection, if the wetland is open to wildlife. More specifically, they declared it technically infeasible to achieve FC consistently <500/100 mL in this situation.

DEVELOPMENTS SINCE 2000

Introduction

Since Schueler's report some additional studies were performed on a variety of BMPs. Constructed wetlands were most commonly investigated in recent years, with the realization that chemicals exuded by plants could be bactericides. Other conventional stormwater BMPs receiving attention were ponds, media filters, vegetated filter strips and swales, and infiltration. There was limited reporting on stormwater disinfection by ultraviolet light.

This review covers each of the types, with the exception of infiltration. If suitable soils and hydrogeologic conditions allow infiltration, it can reduce pollutant inputs to surface waters by 100 percent. However, these conditions are unlikely to be prevalent in Blaine because of the predominance of glacial till soils.

Commercial enterprises have introduced a variety of proprietary BMPs to the market in recent years. The literature reports the success in FC reduction of three types: StormFilter, a media filter; StormTreat, a packaged wetland system, and the Stormceptor and Vortechnics devices, which employ hydrodynamic mechanisms for removing particles by centrifugal or centripetal force.

Constructed Wetlands

Australian researchers studied the bacteria reduction performance of a stormwater constructed wetland, as well as a wet pond (Davies and Bavor 2000; Bavor, Davies, and Sakadevan 2001). The wetland was elongated relative to its width (length:width ratio approximately 7:1) and was planted extensively with *Phragmites australis*. Discrete (presumably, grab) inflow and outflow samples were collected weekly. Mean removal efficiencies for FCs, enterococci, and heterotrophic bacteria were 79, 85, and 87 percent, respectively, with influent concentrations of the order 10^2-10^5 for the first two organism groups and 10^6-10^7 for heterotrophs. The lowest effluent FC concentration was 200/100 mL, well above the Washington shellfish criterion of ≤14/100 mL as a geometric mean.

Bavor, Davies, and Sakadevan (2001) reported on settling experiments, which demonstrated that bacteria were almost exclusively associated with particles less than 2 μm in size. Others (e.g., Dale 1974) noted this tendency of microorganisms to adsorb to particles, especially the finer ones. Wong, Breen, and Somes (1999) observed that bacteria are removed from stormwater principally through sedimentation. The very small particles transporting most of the bacterial load are difficult to settle, but filtering through vegetation assists settling. Once deposited, sediment-bound bacteria still can be resuspended back into the water column through disturbance by subsequent high storm flows (Crabill et al. 1999). Good vegetation cover could again assist performance by stabilizing sediments and reducing perturbation by flow (Davies and Bavor 2000).

Relative performance of a stormwater and a wastewater wetland was compared in Sweden (Stenstrom and Carlander 2001). The stormwater wetland had a sedimentation pond, shallow vegetated zone, and denitrification pond, with an overall water residence time of 3-5 days. The wastewater wetland had two parallel pond systems providing a 7-day residence time. The sampling procedures were not described. The wastewater wetland achieved very high removal efficiencies for E. coli, FC, and *Clostridium* (an anaerobic spore-forming bacterium) in both warmer and cooler seasons (*E. coli*—99.8% May, 97.5% November; FC—99.9% May and

November; *Clostridium*—98.7% May, 95.9% November). The researchers observed a relationship between efficiencies of bacteria and particulate reductions, indicating again bacterial transport with the solids and removal through settling. The stormwater wetland reduced only total coliforms, and those bacteria only by one order of magnitude. However, entering concentrations were already relatively low for urban runoff at 10^2-10^3.

The Swedish research included sediment survival studies in the stormwater wetland. It took 24-27 days and 27-53 days for 90 percent die-off of *E. coli* and enterococci, respectively (and much longer for *Clostridium* and viruses). Thus, pathogens are vulnerable to remobilization by disturbances for a relatively long time.

California Department of Transportation (Caltrans, 2004) comprehensively studied the full range of conventional treatment BMPs, including a constructed wetland, at highway, maintenance station, and park-and-ride sites. Samples for FC analysis were collected as single grabs from the influent and effluent, and removal efficiencies were not computed. Influent concentrations at the constructed wetland, which was within a freeway right of way, ranged from 2 to 50000/100 mL, and at the outlet 2 to 7000/100 mL. The majority (65 percent) of the effluent samples had concentrations of the order 10^1/100 mL. In contrast, discharge concentrations at other BMPs included in the program were 10^2-10^3/100 mL in the majority of cases (see reports under the headings Ponds, Media Filters, and Vegetated Filter Strips and Swales below).

Two Alaska sedimentation basin-constructed wetlands systems receiving highway runoff were monitored during the fall season without description of the sampling scheme (Nyman et al. undated). Fecal coliforms were reduced to less than 10/100mL from already low (but unreported) numbers in the influent. A risk of using constructed wetlands or ponds for treatment of FC contamination is that the open water often attracts water fowl and wildlife, ultimately increasing contamination levels. A team of California researchers found this risk to be real in a constructed saltwater marsh near Huntington Beach. They found that Talbert Marsh regularly flushes millions of gallons of bird droppings into the Pacific Ocean. The research concluded that saltwater marches should be designed to discharge at a slower rate. A slower flow rate would likely prevent most contamination, since longer exposure to salt water and sunlight kills the bacteria (Grant et al. 2001). Any open water treatment facility should be carefully designed with this risk in mind.

StormTreat, a Modular, Manufactured Constructed Wetland

StormTreat is an in-ground modular device 2.9 meters (9.5 ft) in diameter consisting of several chambers manufactured and marketed by StormTreat Systems, Inc. A series of sedimentation chambers at the entrance are constructed to skim floatables (e.g., oils) as well as settle solids. The ultimate chamber is a vegetated wetland planted in gravel, where the water enters at the root zone. StormTreat is intended to treat the first 1.27 cm (0.5 inch) of runoff from relatively small storms or the first flush of larger events. Serving very large areas or attempting to treat larger flows requires a number of parallel units and a complex distribution arrangement. In many situations the standard StormTreat design basis would not comply with the 1992 Washington Department of Ecology designated water quality design storm, the 6-month, 24-hour rainfall event, which is equivalent to approximately 1.4 inch in Blaine. This storm would produce 0.5

inch or less of runoff only if the runoff coefficient were under 0.36. The 2005 Ecology Manual requires effective treatment for 91% of the runoff volume, which is actually less than providing treatment for the 6-month, 24-hour rainfall event. For simplicity, the cursory calculations completed for this memo were based on the 1992 requirements.

Sonstrom, Clausen, and Askew (2002) conducted a thorough study of a StormTreat system treating runoff from a roof and parking lot at a commercial site in Connecticut over a 2-year period. Two parallel units served 0.27 hectare (0.67 acre). This installation could treat only the first 0.46 cm (0.18 inch) of runoff. More tanks would have been necessary to meet the standard design basis, but the site owner would not make available the needed space and budget to do so. Excess runoff bypassed and was not monitored. Therefore, this study does not portray performance in the recommended configuration but does provide data on the device's capabilities when individual units receive the design flow.

Grab sampling of the influent and effluent of the parallel units provided 16 samples for FC analysis. The hydraulic residence time was determined to average 9 days. Accordingly, effluent concentrations were compared to influent concentrations from the preceding week. This study thus made some attempt to compensate for the usual problem of inflows and outflows being from different water volumes.

Over the full Connecticut study the influent had median FC of 12000/100 mL and a mean of 590/100 mL. The effluent mean was < 1/100 mL. The researchers estimated cumulative loading reduction of FC at 99 percent. They attributed the high degree of retention to entrapment, filtration, and die-off.

This StormTreat system was thus shown to be capable of meeting water quality criteria for shellfish at discharge. It must be recalled, though, that it treated only a fraction of the runoff generated by the catchment. Assuming a runoff coefficient of 0.8 for the highly impervious site, it would have taken 13 units to meet the 1992 Washington Department of Ecology's design criterion of treating runoff from 1.4 inch of rainfall.

Other reports of FC reduction in StormTreat systems range from 83 percent (Federal Highway Administration, undated) to 97 percent (StormTreat Systems, Inc, undated). The latter report from the manufacturer's website incorporates data from several client studies verified by a certification program for proprietary BMPs operated by the state of Massachusetts.

Ponds

The Australian research on constructed wetlands reported above also included monitoring of a wet pond (Davies and Bavor 2000; Bavor, Davies, and Sakadevan 2001). A wet pond has a permanent or semi-permanent pool in which water has a relatively long residence time for reduction of small solids and dissolved substances, differing from a constructed wetland in having less or no submerged or emergent vegetation. The Australian pond had three cells, each approximately 2.5 meters (8.2 ft) in depth, with a fringe of *Typha* (cattails). This pond removed little or no bacteria (efficiencies of –2.5, 23, and 22 percent for FC, enterococci, and heterotrophic bacteria, respectively). It was in a watershed undergoing construction and had a significantly higher proportion of particles smaller than 5 μm than did the catchment feeding the wetland.

Mallin et al. (2002) grab sampled the inflow and outflow from three wet ponds receiving urban runoff over a 29-month period and measured FC concentrations. The geometric means declined from 488 to 70/100 mL and 97 to 43/100 mL in two ponds (efficiencies of 86 and 56 percent, respectively) but increased from 74 to 85/100 mL in a pond receiving golf course runoff. Therefore no pond effluent would meet the Washington shellfish criterion of ≤ 14/100 mL as a geometric mean.

A report from the Virgin Islands (Anonymous, undated) recounted comparative FC measurements at the inlet and outlet of a pond through a storm (presumably with grab sampling). Eight inflow samples varied from 18 to 810/100 mL. Mean removal efficiency was 76 percent, but the median was higher at 90 percent. The geometric mean of the effluent concentrations was 41/100 mL, again above the Washington criterion.

The Caltrans (2004) research included extended-detention ponds, which held runoff for up to 72 hours. This residence time is not nearly as long as in a constructed wetland or a wet pond but does offer some enhanced settling. Influent FC concentrations ranged from 110 to 28000/100 mL. Effluents exhibited concentrations ranging from 2 to 90000/100 mL, with the majority of values being of the order 10^2-10^3/100 mL.

Media Filters

The Caltrans (2004) study also encompassed sand filters and a StormFilter unit, which was at a maintenance station. Sand filters were of two types: the "Austin" design, in which flow enters a sedimentation chamber at a single point and then discharges via a perforated riser pipe onto sand; and the "Delaware" design, in which sheet flow enters a sedimentation chamber along a broad flow path and then passes over a weir to the sand chamber. A StormFilter has a bank of canisters containing a filtration medium, in this case perlite-zeolite. It is manufactured and marketed by Stormwater Management, Inc. (now Stormwater360).

Sand filter influent concentrations ranged from 23 to 200000/100 mL, with effluents covering the range 2 to 50000/100 mL. The majority of effluent concentrations were of the order 10^2-10^3/100 mL. Flows in the StormFilter ranged from 8 to 9000/100 mL. The effluent range was 2 to 3000, with 71 percent of the values of the order 10^2-10^3/100 mL.

Stormwater360 believes that subsurface constructed wetlands may be the most cost-effective treatment solution for FC reduction in stormwater. Stormwater360 is in the conceptual stage of a pilot project using the StormFilter in conjunction with subsurface wetlands. This eventual pilot will be in conjunction with Stephen Lyons, Ph.D., P.E., and/or Orange County Water District (Anaheim, CA).

Vegetated Filter Strips and Swales

Casteel et al. (2005) quantified bacterial indicators of fecal contamination in stormwater before and after diversion to a natural vegetated riparian buffer adjacent to a lake in the San Francisco. Lake concentrations of *E. coli*, enterococci, and total coliforms were about two to three orders of magnitude (99-99.9%) lower with treatment in the buffer than levels in stormwater, presumably based on grab sampling.

The Caltrans (2004) research covered both filter strips and swales. Filter strips are broad vegetated slopes receiving sheet flow, while swales are vegetated channels flowing at some depth. Filter strips experienced inflows having FCs from 30 to 90000/100 mL and discharged 17 to 9000/100 mL. The equivalent ranges for swales were 17 to > 200000/100 mL in the inflows and 17 to > 200000/100 mL in the effluents. The majority of effluent concentrations were of the order 10^2-10^3/100 mL for both BMP types.

Stormwater Disinfection

The city of Encinitas, CA studied ozonation and ultraviolet (UV) processes for disinfecting stormwater runoff to protect a swimming beach (Rasmus and Weldon 2005). A preliminary paper assessment rejected ozonation on a variety of logistical, cost, and performance grounds. Monitoring of the selected UV system for three months in the fall of 2002 showed the following reductions in geometric means of daily data: total coliforms—23437 to 2/100 mL, FC—1849 to 2/100 mL, and enterococci—1563 to 2/100 mL. Therefore, UV disinfection can reliably meet water quality criteria, although with considerable difficulty and expense to treat large stormwater volumes.

Hydrodynamic Devices

Neary and Boving (2004) reported on the performance of a Vortechs Stormwater Treatment System, a product of Vortechnics, Inc. (now Stormwater360). Flow enters the unit tangentially to a grit chamber, which promotes a swirling motion driving particles toward the center, where velocities are lowest and some settling occurs. Water then passes under a baffle to separate floatables. Flows above the design quantity bypass the unit. The authors did not describe the sampling procedure for FCs. Their removal ranged from 50% to 88% during three spring sampling events.

Other reports on Vortechs are less encouraging. The net removal was negative as reported in two studies by Clausen et al. (2002) and West et al. (2001).

Stormceptor is another commercial hydrodynamic device from the Stormceptor Group of Companies. Stormwater flows into an upper bypass chamber, where a weir and orifice assembly diverts flows less than the design rate into a lower treatment chamber. Velocity slows when water enters the treatment chamber. Here floatables rise and solids settle by gravity. From the treatment chamber, water is displaced up through a riser pipe into the bypass chamber on the downstream side of the weir for discharge. Clausen et al. (2002) and Waschbusch (1999) studied performance of Stormceptor units and found their net FC removal to be negative.

It was established above that FCs have a strong association with the smallest particles. These hydrodynamic devices have little capability of capturing relatively small particles and function well only in removing large solids like trash and the high end of the particle spectrum.

LOCAL PILOT PROJECTS AND RESEARCH

In 2003-04, the Port of Bellingham, Whatcom County Marine Resources Committee, Whatcom County, City of Blaine and the Drayton Harbor Shellfish Advisory Committee, through a cooperative effort, researched and developed stormwater treatment management practices to reduce bacterial pollution in Blaine Harbor, specifically near the Blaine Marina. The effort resulted in two pilot projects that were developed and implemented; installation of spiders on the breakwater to discourage seagull and pigeon roosting, and stormwater planters at the downspouts of the webhouse roof in the Blaine Marina (Landau Associates, Inc. 2004).

The stormwater planters were designed to "filter" the water for fecal coliform bacteria and other pollutants before the runoff drains into the marina waters. The rain water running off the roof of Webhouse 1 has very high concentrations of these bacteria, likely from the rain washing bird droppings left by the many gulls that regularly roost on the webhouse roof. Rainwater from the roof's downspouts is collected in the stormwater planter where it slowly filters through plant roots, soil and sand. The fecal coliform bacteria are captured in the soils where they break down and get absorbed by the plant roots. Filtered water empties into a storm drain that carries it to the marina (Landau Associates, Inc. 2004).

The stormwater planters were installed in the spring of 2004. Because of funding limitations the planters were not installed as originally specified. In fact, the total planter area was undersized by 85-97%. The projected removal rate for the planters was 99%. Because the system was so severely undersized, several of the monitored events overflowed the planters. Not counting this event, the removal rate was 50% (Hirsch Consulting Services 2004).

Considering that the planters were extremely undersized, the removal rates appear to be promising. The planter box pilot project experienced dead vegetation, possibly as a result of over-fertilization. A key recommendation included in the monitoring report of the stormwater planters suggested specifying plants that can tolerate high organic loading (Hirsch Consulting Services 2004). This recommendation should be considered with the construction of any treatment facility that includes vegetation as part of the treatment, such as wetlands and the StormTreat system. Using the stormwater planter technology on a much larger scale may be a feasible option within the City of Blaine.

SUMMARY AND CONCLUSIONS

Urban stormwater runoff is a widely recognized source of shellfish contamination by potential disease-causing organisms, which can lead to the closure of beds to harvest for human consumption. The fecal coliform group of bacteria is a convenient indicator of disease potential associated with a variety of microorganisms. FCs do have several disadvantages associated with their broad range of extra-human sources, lack of uniform association with pathogens, variability, and monitoring difficulties. Nevertheless, no better alternative has yet emerged, and FCs are used as the basis for assessing shellfish bed status.

The State of Washington sets as water quality criteria for shellfish waters a geometric mean of FC readings not to exceed 14/100 mL, with no more than 10 percent of the measurements surpassing 43/100 mL. Therefore, stormwater discharge targets should be of the order 10^1/100 mL, unless great dilution of the discharge can be assured.

Two general methods exist to prevent or reduce shellfish bed contamination by urban stormwater: pollution source controls and runoff treatment. Source controls separate the points of pollution origin from contact with rainfall or runoff; if the separation is complete, they are 100 percent effective in preventing contamination. Runoff treatments attempt to remove pollutants already in runoff; they can reduce but cannot entirely prevent contamination, unless all runoff infiltrates the soil and only emerges to surface water after full pathogen die-off.

This literature review investigated commonly used urban stormwater treatment techniques: constructed wetlands, ponds, media filters, vegetated filter strips and swales, and hydrodynamic devices. It also covered the small amount of information available on stormwater disinfection.

Excluding disinfection, constructed wetlands yielded the best performance in terms of fecal coliform reduction efficiency and effluent quality. All other options reviewed, except disinfection, generally produced effluents with FC concentrations two to three orders of magnitude higher than the presumed target of ~10^1/100 mL. Ultraviolet disinfection has been shown, as would be expected, to lower concentrations below detection. While this option could receive more consideration by the City of Blaine, it is likely to prove too logistically difficult and expensive for widespread application to protect shellfish beds.

Even with constructed wetlands, effluent FC concentrations were still generally an order of magnitude above the ~10^1/100 mL target. The major exception to this observation was the StormTreat system, a modular, manufactured constructed wetland on the commercial market, which reduced influent concentrations ranging 10^2-10^4/100 mL to a mean below detection.

Kadlec and Knight (1996), in evaluating results from municipal wastewater treatment in wetlands, offered an important clue regarding why the StormTreat system can out-perform large, more naturalistic constructed wetlands in FC reduction. They concluded that constructed wetland outflow concentrations cannot consistently be reduced to near zero, or even close, without disinfection, if the wetland is open to wildlife. This point was also illustrated in the research of Grant et al (2001) on the man-made Talbert Marsh, concluding that the additional seagull droppings were a direct source of FCs in the surf zone along Huntington Beach. The StormTreat units are not conducive to wildlife occupancy or access by domestic animals. The Caltrans (2004) experience with a constructed wetland in an urban freeway right of way adds evidence supporting this conclusion. This wetland was not easily accessible or attractive to wildlife and domestic animals. It exhibited the lowest effluent concentrations among the installations reviewed, although they were still considerably above the StormTreat levels.

The StormTreat system thus has potential for serious further consideration by the City of Blaine from the performance standpoint. However, treatment of the full State of Washington water quality design storm would require multiple units on all but the smallest sites, with the attendant issues of space, hydraulics, and cost.

More broadly, the City should investigate other ways to use constructed wetland technology while excluding animals that excrete fecal coliforms. There are models of wetland configurations from the municipal treatment experience that have not been investigated enough, if at all, for stormwater treatment, particularly the subsurface-flow class of constructed wetlands. These wetlands differ from the usual type used in stormwater management and reviewed here by having an artificial growth medium in a geometrically regular, constructed chamber with the water level at or below the medium surface, and often a surrounding fence. In other words, they are built like a wastewater treatment system, with little to attract animals. In contrast, the usual stormwater constructed wetlands have open water pools and emergent plant zones, a natural soil substrate, irregular shape, and open access. In other words, they are built somewhat like a natural water body and attract at least urban animals.

Coupled with further investigation of proprietary and non-proprietary constructed wetland designs, the City of Blaine should catalogue and assess every possible source control strategy that might be used to reduce initial FC concentrations in stormwater runoff to the minimum possible. Implementing the best feasible source controls would not replace the need for treatment but would add assurance to its success.

REFERENCES

Anonymous. Undated. Summary Report on Fecal Coliform Bacteria Removal Efficiency for Stormwater Runoff BMPs in the Virgin Islands. http://water.usgs.gov/wrri/02-03grants_new/prog-compl-reports/2003VI9B.pdf.

Bavor, H.J., C.M. Davies, and K. Sakadevan. 2001. Stormwater treatment: Do constructed wetlands yield improved pollutant management performance over a detention pond system? *Water Science and Technology* 44(11-12):565-570.

California Department of Transportation. 2004. BMP Retrofit Pilot Program Final Report (Appendix F), CTSW-RT-01-050. California Department of Transportation, Sacramento, CA.

Casteel, M.J., G. Bartow, S.R. Taylor and P. Sweetland. 2005. Removal of Bacterial Indicators of Fecal Contamination in Urban Stormwater Using a Natural Riparian Buffer. 10th International Conference on Urban Drainage, Copenhagen/Denmark, 21-26 August 2005. http://66.102.7.104/search?q=cache:XnPKgPSYSQkJ:www.lakemerced.org/Plans/Water/VistaGrande/Casteeletal_10icud_paper.PDF.

Clausen, J.C., P. Belanger, S. Board, M. Dietz, D. Phillips, and R. Sonstrom. 2002. Final Report, Stormwater Treatment Devices, Section 319 Project. Connecticut Department of Environmental Protection, Hartford, CT.

Crabill, C., R. Donald, J. Snelling, R. Foust, and G. Southam. 1999. The impact of sediment fecal coliform reservoirs on seasonal water quality in Oak Creek, Arizona. Water Research 33:2163-2171.

Dale, N.G. 1974. Bacteria in intertidal sediments: Factors related to their distribution. Limnology and Oceanography 19:509-518.

Davies, C.M. and H.J. Bavor. 2000. The fate of stormwater-associated bacteria in constructed wetland and water pollution control pond systems. Journal of Applied Microbiology 89:349-360.

Federal Highway Administration. Undated. http://www.fhwa.dot.gov/environment/ultraurb/uubmp3p9.htm.

Grant, S.B., B.F. Sanders, A.B. Boehm, J.A. Redman, J.H. Kim, R.D. Mike, A.K. Chu, C.D. Gouldin, C.D. McGee, N.A. Gardiner, B.H. Jones, J. Sveikovsky, G.V. Leipzig, and A. Brown. 2001. Generation of enterococci bacteria in a coastal saltwater marsh and its impact on surf zone water quality. Environmental Science and Technology 35: 2407–2416.

Hirsch Consulting Services. 2004. Blaine Harbor Stormwater Treatment Pilot Monitoring. Prepared for the Port of Bellingham by Hirsch Consulting Services, Lummi Island, WA.

Kadlec, R.H. and R.L. Knight. 1996. Treatment Wetlands. Lewis Publishers, Boca Raton, FL.

Landau Associates, Inc. 2004. Specification for Stormwater Filter/Planter. Prepared for the Port of Bellingham by Landau Associates, Inc., Edmonds, WA.

Mallin, M.A., S.H. Ensign, T.L. Wheeler, and D.B. Mayes. 2002. Pollutant removal efficacy of three wet detention ponds. Journal of Environmental Quality 31(2):654-660.

Neary, K.A. and T.B. Boving. 2004. Stormwater Runoff Treatment with Vortechs® Structures. Geological Society of America Annual Meeting, Denver, CO, November 7–10, 2004. http://gsa.confex.com/gsa/2004AM/finalprogram/abstract_77164.htm.

Nyman, D., C. McCauley, D. Maddux, W. Schnabel, C. Woolard, and C. Adler. Undated. Evaluation of Stormwater Treatment by Constructed Wetlands: A Study of Two Sedimentation Basin/Constructed Wetlands at Soldotna, Alaska. www.awra.org/state/alaska/ameetings/2003am/abstracts/Wed-Nyman-wtlds.doc.

Pitt, R. 1998. Epidemiology and stormwater quality management. In Stormwater Quality Management, CRC/Lewis Publishers, New York, NY.

Rasmus, J. and K. Weldon. 2005. Moonlight Beach urban runoff treatment facility: Using ultraviolet disinfection to reduce bacteria counts. Stormwater, March 2005.

Schueler, T.R. 1999a. Microbes in urban watersheds: Concentrations, sources, and pathways. Watershed Protection Techniques 3(1):554-565.

Schueler, T.R. 1999b. Microbes in urban watersheds: Ways to kill 'em. Watershed Protection Techniques 3(1):566-574.

Sonstrom, R.W., J.C. Clausen, and D.R. Askew. 2002. Treatment of parking lot stormwater using StormTreat system. Environmental Science and Technology 36:4441-4446.

Stenstrom, T.A. and A. Carlander. 2001. Occurrence and die-off of indicator organisms in the sediment in two constructed wetlands, Water Science and Technology 44(11-12):223-230.

StormTreat Systems, Inc. Undated. http://www.stormtreat.com/stsdata.html.

Waschbusch, R.J. 1999. Evaluation of the Effectiveness of an Urban Stormwater Treatment Unit in Madison, Wisconsin, 1996-1999. USGS Water Resources Investigation Report 99-4195. U.S. Geological Survey, Middleton, WI.

West, T.A., J.W. Sutherland, J.A. Bloomfield, and D.W. Lake. 2001. A Study of the Effectiveness of a Vortechs Stormwater Treatment System for Removal of Total Suspended Solids and Other Pollutants in the Marine Village Watershed, Village of Lake George, New York. New York Department of Environmental Conservation, Albany, NY.

Wong, T.F.H., P.F. Breen, and N.L.G. Somes. 1999. Ponds Vs. Wetlands—Performance Considerations in Stormwater Quality Management. Proceedings of the Comprehensive Stormwater and Aquatic Ecosystem Management First South Pacific Conference, 22-26 February 1999, Auckland, New Zealand, pp. 223-231.

Appendix 4H: Summary of Protection and Restoration Strategies for Watersheds and Tributaries (Section 3)

Following is a compilation of all strategies identified in Section 3 for the protection and restoration of watersheds and tributaries. See Appendix 4H, Table H1 for a summary of references with more information, threats addressed, and relationships with PSP's Results Chain strategies.

Key Strategy (4A): Develop a comprehensive watershed-based management system.

Key Strategy (4B): Manage stream watersheds using a data- and objective-based approach with appropriate specific strategies for streams depending on their levels of ecological condition.

Key Strategy (4C): Synthesis of guiding principles for stream restoration

- Protect well functioning streams and their habitats, where they exist.
- Consider what actions are necessary in the contributing watershed to achieve restoration goals and objectives. Either take these actions according the Strategy 4B or, if they cannot be performed, adjust goals and objectives to what is attainable or transfer restoration activity to a location where they can.

- Identify in-stream restoration options and apply the hierarchical strategy of Roni et al. (2002) to prioritize among them. That strategy emphasizes habitat reconnection as generally the most effective and certain of in-stream strategies, where prior disconnection is among the problems. The strategy then guides a user through consideration of riparian restoration and road improvements, with in-stream structural placements to follow or occur simultaneously with any of the other actions, as appropriate.

Key Strategy (4D): Protect, restore, and create wetlands according to the known preferences and tolerances of target biological communities, particularly geomorphic, hydrological, and hydroperiod requirements.

Key Strategy (4E): Protect and restore lakes applying the established specific strategies of algal biomass and macrophyte control.

Key Strategy (4F): As the principal basis of urban stormwater management, apply Aquatic Resources Conservation Design practices in a decentralized (i.e., close to the source), integrated fashion to new developments, redevelopments, and as retrofits in existing developments as necessary to meet established protection and restoration objectives. If a full, scientifically based analysis shows that it is indeed impossible to meet objectives with these practices, employ, first, in lieu fees or trading credits or, as a second priority option, conventional stormwater management practices according to the following key strategy:

Key Strategy (4G): Employ conventional stormwater management practices when the above options do not fully meet objectives. Increase the effectiveness of conventional vegetation- and soil-based practices whenever possible by using ARCD landscaping techniques. Apply enhanced filtration, ion exchange, or a treatment train involving both in industrial situations when source controls and ARCD measures are insufficient to meet objectives.

Key Strategy (4H): Address special stormwater problems as follows A. Promote source control under a broad ARCD program by assessing ubiquitous, bioaccumulative, and/or persistent pollutants that can only be controlled well by substituting with non-polluting products and enact bans on the use of products containing those pollutants. B. Improve construction site stormwater control by prioritizing, first, construction management practices that prevent erosion and other construction pollutant problems; second, practices that minimize erosion; and, last, sediment collection after erosion has occurred. C. To counteract dispersed sources of pathogens that compromise shellfish production and other beneficial uses, implement strong source controls and treat remaining sources with subsurface-flow constructed wetlands, assuming additional research and development verifies the promise of that technique.

Key Strategy (4I): Bolster incomplete combined sewer overflow reduction programs by using ARCD techniques identified for application in that setting to decrease stormwater flows.

Key Strategy (4J): If nitrogen discharge from a municipal treatment plant must be reduced below 1 mg total nitrogen/L to remove a threat to marine dissolved oxygen resources, apply reverse osmosis tertiary treatment with highly efficient filtration as a pretreatment. If analysis demonstrates that a lesser reduction will suffice, apply membrane bioreactor treatment. Key

Strategy(4K): If discharges from on-site wastewater treatment systems are a serious threat to: (1) marine dissolved oxygen resources as a result of nitrogen; or (2) shellfish production or contact recreation as a result of pathogens, assess as possible solutions: (1) construct sewers and a municipal treatment plant, with advanced treatment for nitrogen if that is the threat, to replace problem on-site systems; or (2) apply advanced on-site treatment, tested and verified to reduce the problem sufficiently to remove the threat (note: at this point more testing is required for both on-site nitrogen removal systems and small-scale disinfection).

Key Strategy (4L): Upgrade the implementation of established agricultural best management practices, especially where agricultural runoff is: (1) a eutrophication threat as a result of nitrogen (N) and/or phosphorus (P); or (2) a threat to shellfish production or contact recreation as a result of pathogens. Manage nitrogen and phosphorus in concert by: (1) employing a phosphorus index to target management of critical P source areas, generally near receiving waters; and (2) applying N-based management to all other areas. Maintenance of riparian buffers advances both facets of the strategy by keeping agricultural activities out of the potentially most critical P production area and providing a sink for N to capture the majority of it before it can enter the water.

Key Strategy (4M): Upgrade the implementation of established forestry best management practices to protect stream water quality and hydrology in the vicinity of forestry activities and minimize the delivery of pollutants from those activities to downstream receiving waters, including Puget Sound.

Table H1. Summary of Protection and Restoration Strategies for Watersheds and Tributaries (Section 4-2)

Key Strategy	Report Reference for Details	Principal Guidance References	Applications	Threats Addressed	Results Chain Strategies Addressed
4A	Appendix 4A, Box A1	DeBarry (2004); Heathcote (2009); NRC (2009); for forestry issues Brooks et al. (2003)	Protection, restoration	All threats originating in Puget Sound watersheds, including those of urban, agricultural, forestry, and rural residential origin	See Appendix 4A, Box A1
4B	Appendix 4B, Table B1	Booth et al. (2001); Horner, May, and Livingston (2003)	Protection (of existing level of biological integrity), restoration (to improve biological integrity from a reduced level)	Stream channel hydromodification; salmon spawning and rearing habitat degradation; stream food web disruption; acute and chronic toxicity effects on aquatic organisms from metal and organic pollutants; increased pollutant loadings to all downstream waters, including Puget Sound	See Appendix 4B, Table B1
4C	Section 4-2 discussion under the heading Effectiveness and Relative Certainty of Stream Restoration	FHWA, (2007); Montgomery et al. (2003); NRCS. (2007a); Roni et al. (2002); Saldi-Caromile et al. (2004); Stewart-Kloster et al. (2009); WDFW. (2003)	Restoration	Restriction of anadromous fish passage; salmon spawning and rearing habitat degradation; stream food web disruption; if watershed restoration involved, threats under Strategy 4B also addressed	RC2 A3; RC4 B1, specifically B1(1), B1(3), and B1(4)
4D	Section 4-2, discussion under the heading Effectiveness and Relative Certainty of Wetlands Management Efforts	Azous and Horner (2001); Granger et al. (2005); Mitsch and Gosselink (2007); Mitsch et al. (2009); Sheldon et al. (2005);	Protection, restoration	Threats associated with their functions, not only to their internal ecosystems but also to waters and terrestrial environments associated with them	A broad range of strategies in this column, because of association of wetlands with other waters
4E	Section 4-2, discussion under the heading Strategies for Management of Lakes	Cooke et al. (2005); Welch and Jacoby (2004)	Protection, restoration	Eutrophication impacts to beneficial uses	No specific strategy
4F	Table 5	Geosyntec Consultants (2008); Hinman (2005); USEPA (2007b); WDOE (2005) Volume IV	Protection (new development), restoration (redevelopment and retrofit)	See Strategy 4B	RC2 A3, specifically A3.3.2; RC6 C2, specifically C2(3), C2(4), C2(6)
4G	Table 12	WDOE (2005) Volumes III, V	Protection (new development), restoration (redevelopment and retrofit)	See Strategy4B	See Strategy 4F
4H-A	Section 4-2	NRC (2009)	Protection,	Acute and chronic toxicity effects on	RC6 C2, RC 7

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Appendix 4I: Research and development needs for implementation of protection and restoration strategies

Here we enumerate the major tasks foreseen by the authors as needed to bring the recommended strategies to full fruition. In some cases these tasks involve research in scientific, technical, or policy arenas; i.e., a systematic inquiry into a subject to discover facts or principles. In other cases the tasks would be more developmental, in the sense of bringing a known method or process to a more advanced or effective state. These research and development (R and D) needs are aligned with the distinct strategies identified in each chapter. Please see the relevant chapter for the citations repeated here.

Research and development needs for implementation of Overarching, Large-Scale Protection and Restoration Strategies

Most of the "Synthesizing Guidance for Puget Sound Protection and Restoration Strategies" relies on basic principles of ecology or well-established scientific findings in the Puget Sound region. Nevertheless, it would be highly valuable to determine the likely gross-scale impacts on key indicators for the Puget Sound ecosystem from different allocations of population growth across the region (i.e., as opposed to the county-by-county projections used for allocating population growth under the Growth Management Act). This could potentially take advantage of the watershed characterizations currently being completed by the Washington Departments of Ecology and Fish and Wildlife, applying them across WRIAs instead of strictly within WRIAs to determine at a regional scale the highest priority locations for protection and restoration and where new development would likely have the least impact. To the extent possible, this analysis should integrate anticipated impacts of climate change, which differ in their scope and severity across the region.

The field of ecological economics asserts that, instead of attempting to calculate the "correct" value of negative or positive environmental externalities, we should act on our knowledge that zero is incorrect. Accepting this challenge, the key research and development need is a feasibility assessment of candidate taxes or fees. The Puget Sound Partnership could choose candidates from potential taxes or fees identified in the Action Agenda and Chapter 4-1.

Research and development needs for implementation of protection and restoration strategies for watersheds and tributaries

Fully implementing the identified protection and restoration strategies for watersheds and tributaries requires a mix of scientific, technical, and institutional research and development activities, as follows.

Key Strategy: Develop a comprehensive watershed-based management system.

- Develop a municipal co-permittee system to manage an integrated set of water-based permits, with a lead permittee working in partnership with other municipalities in the watershed as co-permittees.

- Establish state and municipal partnerships by watershed to set goals and objectives for protection and restoration, according to the principles outlined in Section 4-2.
- Establish a highly professional structure to perform the scientifically and technically based watershed analyses necessary to set and achieve goals and objectives.
- Set up the legal, regulatory, and financing mechanisms as necessary to assign authority and responsibility to municipal co-permittees for achieving goals and objectives and to ensure adequate funding for doing so.
- Determine the extent of institutional and financial barriers to retrofitting watersheds with stormwater and wastewater infrastructure necessary to meet goals and objectives and how they can be overcome.
- Develop an in lieu fee and credit trading system to make it possible for development project sponsors to compensate for legitimate inability to meet requirements on-site by supporting equivalent effort elsewhere within the same watershed.
- Incorporate recommended monitoring strategies into the monitoring program development efforts proceeding separately from the Puget Sound Science Update.

Key Strategy: Manage stream watersheds using a data- and objective-based approach with appropriate specific strategies for streams depending on their levels of ecological condition.

- Develop the watershed databases necessary to perform the recommended assessments.

Key Strategy: Restore streams according to a set of following principles given in Section 4-2.

- Adapt for urban application the hierarchical strategy for prioritizing restoration developed by Roni et al. (2002).

Key Strategy: Protect, restore, and create wetlands according to the known preferences and tolerances of target biological communities, particularly geomorphic, hydrological, and hydroperiod requirements.

- Determine the barriers that have impeded the application of knowledge about preferences and tolerances of target biological communities in wetland mitigation projects and act to remove them.

Key Strategy: Protect and restore lakes applying the established specific strategies of algal biomass and macrophyte control.

- No additional R and D required.

Key Strategy: As the principal basis of urban stormwater management, apply Aquatic Resources Conservation Design (ARCD) practices in a decentralized (i.e., close to the source), integrated fashion to new developments, redevelopments, and as retrofits in existing developments as necessary to meet established protection and restoration objectives. If a full, scientifically based analysis shows that it is indeed impossible to meet objectives with these practices, employ, first, in lieu fees or trading credits or, as a second priority option, conventional stormwater management practices according to next key strategy.

- Perform research to make objective determinations of the pavement widths actually needed for streets with various service levels and other paved areas.
- Determine how best to move the construction industry to act in such a way that soil disturbance is minimized during construction.
- Perform research to determine the best techniques for maximizing evapotranspiration (ET) from ARCD facilities, and the contribution ET can make in the Puget Sound region to reducing surface runoff from developed areas.
- Perform research to determine the best soil amendment techniques (composition and quantity) for maximizing soil storage, infiltration, and ET in ARCD facilities.
- Perform research on the various permeable pavement types to determine how best to extend their life both structurally and hydrologically.
- Perform research to determine the best vegetated-roof design techniques to maximize storage and ET, and the contribution green roofs can make in the Puget Sound region to reducing surface runoff from developed areas.
- Perform research to determine how much building with full ARCD application can be allowed, starting from different levels of existing development, and still prevent deterioration of biological integrity below existing levels in waters receiving storm runoff.

Key Strategy: Employ conventional stormwater management practices when the above options do not fully meet objectives. Increase the effectiveness of conventional vegetation- and soil-based practices whenever possible by using ARCD landscaping techniques. Apply enhanced filtration, ion exchange, or a treatment train involving both in industrial situations when source controls and ARCD measures are insufficient to meet objectives.

- Perform research to determine the benefits of applying ARCD landscaping principles and methods in vegetation- and soil-based conventional stormwater facilities.

Key Strategy: Address special stormwater problems as follows:

A. Promote source control under a broad ARCD program by assessing ubiquitous, bioaccumulative, and/or persistent pollutants that can only be controlled well by substituting with non-polluting products and enact bans on the use of products containing those pollutants.

- Catalogue ubiquitous, bioaccumulative, and persistent pollutants threatening the Puget Sound ecosystem, less threatening alternatives already available, and cases where development of such alternatives is needed to make substitutions.
- Develop legal, legislative, and regulatory structures for banning threatening chemicals in relation to alternative availability.

B. Improve construction site stormwater control by prioritizing, first, construction management practices that prevent erosion and other construction pollutant problems; second, practices that minimize erosion; and, last, sediment collection after erosion has occurred.

- No additional R and D needed.

C. To counteract dispersed sources of pathogens that compromise shellfish production and other beneficial uses, implement strong source controls and treat remaining sources with subsurface-flow constructed wetlands, assuming additional research and development verifies the promise of that technique.

- Test subsurface flow wetlands, designed to exclude wildlife, for pathogen reduction in stormwater runoff and develop design and maintenance specifications that provide maximum reduction.

Key Strategy: Bolster incomplete combined sewer overflow reduction programs by using ARCD techniques identified for application in that setting to decrease stormwater flows.

- No additional R and D required.

Key Strategy: If nitrogen discharge from a municipal treatment plant must be reduced below 1 mg total nitrogen/L to remove a threat to marine dissolved oxygen resources, apply reverse osmosis tertiary treatment with highly efficient filtration as a pretreatment. If analysis demonstrates that a lesser reduction will suffice, apply membrane bioreactor treatment.

- Perform research to determine the level of municipal wastewater nitrogen reduction required to protect marine dissolved oxygen resources in specific cases.
- If reverse osmosis is required for protection in at least some cases, perform research to determine if its cost can be reduced sufficiently to improve its cost-effectiveness substantially.

Key Strategy: If discharges from on-site wastewater treatment systems are a serious threat to: (1) marine dissolved oxygen resources as a result of nitrogen; or (2) shellfish production or contact recreation as a result of pathogens, assess as possible solutions: (1) construct sewers and a municipal treatment plant, with advanced treatment for nitrogen if that is the threat, to replace problem on-site systems; or (2) apply advanced on-site treatment, tested and verified to reduce the problem sufficiently to remove the threat (note: at this point more testing is required for both on-site nitrogen removal systems and small-scale disinfection).

- Thoroughly test promising on-site nitrogen removal technologies under Puget Sound conditions to determine if such a system can reduce nitrogen sufficiently to protect marine dissolved oxygen resources in specific cases where they are threatened by on-site treatment system discharges.
- Further develop small-scale disinfection technologies to improve their cost-effectiveness.

Key Strategy: Upgrade the implementation of established agricultural best management practices, especially where agricultural runoff is: (1) a eutrophication threat as a result of nitrogen (N) and/or phosphorus (P); or (2) a threat to shellfish production or contact recreation as a result of pathogens. Manage nitrogen and phosphorus in concert by: (1) employing a phosphorus index to target management of critical P source areas, generally near receiving waters; and (2) applying N-based management to all other areas. Maintenance of riparian buffers advances both facets of the strategy by keeping agricultural activities out of the potentially most critical P production area and providing a sink for N to capture the majority of it before it can enter the water.

- Develop the framework to institutionalize this strategy in watersheds subject to the negative impacts of eutrophication and, in general, to provide more directed guidance on the full range of contaminant issues to Puget Sound agricultural concerns.

Key Strategy: Upgrade the implementation of established forestry best management practices to protect stream water quality and hydrology in the vicinity of forestry activities and minimize the delivery of pollutants from those activities to downstream receiving waters, including Puget Sound.

- Reinvigorate the Timber Fish Wildlife process to implement this strategy in a strong partnership with the Puget Sound Partnership.

Research and development needs for implementation of Marine and Estuarine Protection and Restoration Strategies 1. Expand and improve our understanding of the sources, pathways, quantities, and fate of pollutants (nutrients, pathogens and toxics) in Puget Sound estuaries and marine waters. Determine how and where they are introduced into estuaries and Puget Sound waters.

2. Determine the effects of priority pollutants on aquatic species and human health. What are the ecological effects of "legacy toxics" such as PCBs and DDT?

3. Identify adaptive mechanisms at organism, population, and community levels that buffer (i.e., reduce vulnerability and promote recovery) the deleterious effects of pollutants.

4. Improve knowledge of times and places ("hotspots") where water quality and sediment are impaired to the point that aquatic biota and/or humans are at risk.

5. What are the times of the year and associated conditions when estuary and marine ecosystems are most at risk?

6. What physical processes affect the distribution and potency of pollutants over time and space?

7. Identify the primary processes affecting the vulnerability and resiliency of PS to perturbation.

8. What effects will climate change have on these processes in the future?

9. Identify areas where the natural and human systems are not integrated, are particularly sensitive to perturbation, or are prone to dysfunction.

10. Eliminate gaps in knowledge and/or uncertainty by conducting research, including controlled, large-scale experiments, modeling and monitoring.

11. What strategies do we recommend to deal with unexpected developments, including catastrophic events?

12. Evaluate the relative effectiveness of current regulatory programs in protecting estuaries and marine areas and mitigating the impacts of human activities.

13. Evaluate the effects of increasing human-caused variation (frequency, amplitude, rates, etc.) in physical conditions (suspended sediment, salinity, etc.) on ecological processes and components.

14. What is the "lag time" between implementation of protection and restoration measures and the expected beneficial effects? What affects the time it takes for ecosystem response and recovery?

15. Develop a comprehensive "data gaps and uncertainties" matrix; update it regularly to ensure that resources are expended where most needed.

16. What are the cumulative effects of bulkheads, docks, piers, etc.?

Research and development needs for implementation of Fisheries and Wildlife Protection and Restoration Strategies

Much research has been conducted on fish and wildlife, particularly salmon, waterflow, and marine mammals. However, in terms of protection and restoration effectiveness, there are still a number of unknowns that need to be addressed. They generally fall into the following categories:

- Dynamic relationships between habitat changes, natural variation, and species' population ecology
- Effects of direct human disturbance on species' behavior (e.g., cetaceans, seabirds, waterfowl)
- Lethal and chronic sub-lethal effects of known and suspected pollutants, (e.g., copper, lead, nano-toxins, surfactants, personal care products, pharmaceuticals, etc.)
- Harvest management (salmon, waterfowl, and shellfish)
- Hatchery management (genetics, competition, mixed-stock fisheries, etc.)
- Effects of ambient light and noise on fish and wildlife behavior
- Quantification of illegal and undocumented harvest