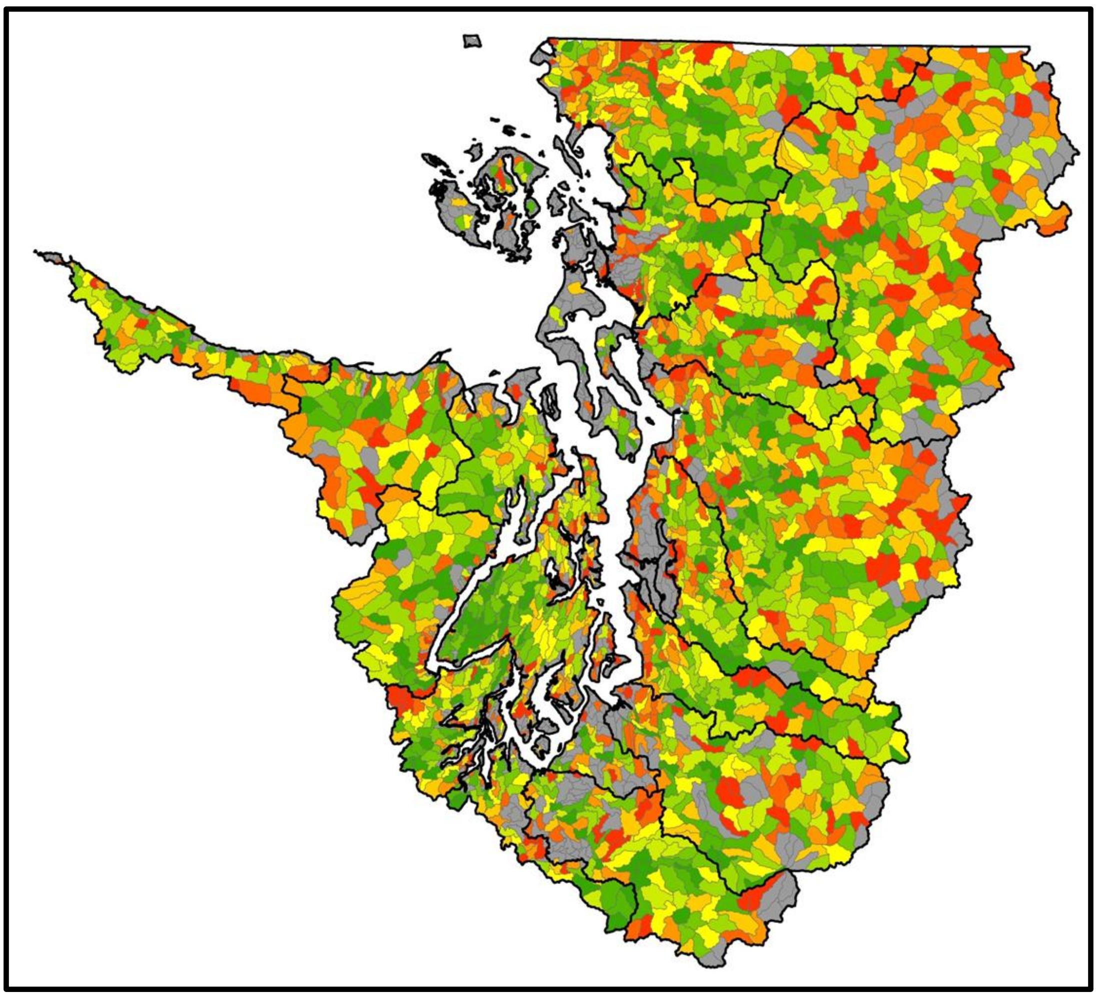
The Puget Sound
Watershed Characterization Project
Volume 2

A Coarse-scale Assessment of the Relative Value of Small Drainage Areas and Marine Shorelines for the Conservation of Fish and Wildlife Habitats in Puget Sound Basin



December 2013

Washington Department of FISH and WILDLIFE

Acknowledgments
Dan Miller, the principal consultant at M2 Environmental Services in Seattle, provided invaluable assistance with some of the GIS analyses done for the freshwater assessment.

Spatial data on the current and historic locations of oak woodland and prairie were provided by the Washington Natural Heritage Program. Susan Grigsby provided spatial data for the assessment units. Ken Pierce of WDFW provided high-resolution classified land cover data for the terrestrial assessment. John Pierce of WDFW, and Joe Rocchio and Rex Crawford of the Washington Natural Heritage Program reviewed portions of the terrestrial habitats assessment and provided insightful advice. John Gamon of the Washington Natural Heritage Program, and John Marzluff and John Withey of the University of Washington reviewed the draft report for the terrestrial habitats assessment and submitted written comments.

Mara Zimmerman, Kirk Krueger, and Ken Warheit of WDFW, Shallin Busch and George Pess of the NOAA Northwest Fisheries Science Center, Bob Bilby of the Weyerhaeuser Company, and Ken Currens of the Northwest Indian Fisheries Commission provided insightful advice at key points in the development of the freshwater habitats assessment. Shallin Busch, George Pess, and Ken Currens reviewed the draft report for the freshwater habitats assessment and submitted written comments.

Scott Pearson and Dayv Lowry of WDFW, Curtis Tanner of U.S. Fish and Wildlife Service, Paul Cereghino of National Marine Fisheries Service, Helen Berry of the Washington Department of Natural Resources, Hugh Shipman of the Department of Ecology, Terry Wright of the Northwest Indian Fisheries Commission, and Charles "Si" Simenstad of the University of Washington reviewed portions of the marine shoreline habitats assessment and provided insightful advice. Scott Campbell of the U.S. Army Corps of Engineers provided assistance and advice for processing the PSNERP GIS data. Kevin Samson and John Jacobson, both of WDFW, did the initial processing of the PSNERP GIS data. Ken Pierce of WDFW routed the marine shoreline segments. Ron Thom and Val Cullinan of the Pacific Northwest National Laboratory, Eric Beamer at the Skagit River System Cooperative, Megan Dethier at the University of Washington, and Paul Cereghino reviewed the draft report for the marine shoreline habitats assessment and submitted written comments.

This work was funded in part by the U.S. Environmental Protection Agency - Aquatic Ecosystems Unit, Wetland Program Development Grant No. CE-96074401-2 which was awarded to the Department of Ecology.

Publication and Contact Information

For more information contact:

George Wilhere
Habitat Program
Washington Department of Fish and Wildlife
Olympia, WA 98501
e-mail: george.wilhere@dfw.wa.gov
Phone: 360-902-2369

Editor's Note: The first edition of this report was published in February 2013. This second edition revises calculations in the freshwater habitats assessment, corrects some errors in the numbering of tables, corrects a figure in Appendix C, adds pseudo-code for both terrestrial and aquatic ecological integrity, and adds data dictionaries for the GIS layers containing the assessments' results in Appendix F.

Preferred citation:

Wilhere, G.F., T. Quinn, D. Gombert, J. Jacobson, and A. Weiss. 2013. A Coarse-scale Assessment of the Relative Value of Small Drainage Areas and Marine Shorelines for the Conservation of Fish and Wildlife Habitats in Puget Sound Basin. Washington Department Fish and Wildlife, Habitat Program, Olympia, Washington.

Table of Contents

Appendices

Executive Summary

Over the next two decades over 1 million additional people are expected to inhabit the Puget Sound Basin. Thousands of acres of agricultural and timber lands will be converted to residential and commercial uses in order to accommodate this phenomenal growth. In addition to providing valuable commodities these "working lands", both agricultural and timber lands, provide habitats for fish and wildlife. Over the coming decades, choices regarding future growth, i.e., choices about where working lands can be converted to new residential development, will have significant effects on the health of many fish and wildlife populations in the Puget Sound Basin.

To ensure the health and well-being of their citizens, promote orderly and efficient land use, and protect natural resources, city and county governments implement comprehensive plans and regulatory land use zoning. Land use zoning largely determines where agricultural and timber lands will be converted to new residential development. To fully realize smart growth, comprehensive land use plans must be based on scientifically-credible information that indicates the most important areas for the conservation of fish and wildlife habitats – areas where new development should be avoided. Our purpose is to provide useful, scientifically-credible information for smart growth in the Puget Sound Basin. Our task is to assess the relative value of places throughout the Basin for the conservation of fish and wildlife habitats.

The Puget Sound Watershed Characterization is a set of spatially-explicit assessments that provide information for regional, county, and watershed-based planning. It is a coarse-scale decision-support tool that should lead to better decisions regarding land use and more effective conservation of our region's natural resources. The assessments cover water resources − both water flow and water quality − and fish and wildlife habitats – in terrestrial, freshwater, and marine shoreline environments – within the Puget Sound Basin. The main products of the assessments are maps that show the *relative* value of small watersheds or marine shorelines throughout the Basin. Relative value is expressed through quantitative indices which can be used to rank places within a water resource inventory area (WRIA) or a county. The indices and the data used to calculate them are stored in a geographic database. The Department of Ecology has led the assessments for water resources and the Department of Fish and Wildlife has led the assessments for habitats. This volume describes the terrestrial, freshwater, and marine shoreline habitats assessments. Refer to Stanley et al. (2011) for descriptions of the water flow and water quality assessments.

Conceptual Foundations

Our approach for assessing relative value is the calculation of indices. An index reduces a complex, multi-dimensional system down to a single number. The resulting simplification facilitates planning and policy making. The Dow Jones Industrial Average, for example, is a stock market index that tracks the day-to-day progress of a highly complex economic system with only a single number that is recalculated each business day. The Dow Jones is intended to provide a big-picture view of the industrial sector of the economy over time. The Dow Jones cannot be used to judge the performance of any particular industry. To gain a better understanding of how various industries have performed, one must examine the many components of the Dow Jones or look to other industry-specific information. Likewise, our indices provide a big-picture view of relative conservation value over the landscape of an entire WRIA or county. They cannot be used to understand the status of particular species or habitats or to design site-level projects.

The relative value of a place for the conservation of fish and wildlife habitats can be based on a variety of different factors: the presence of rare species or habitat-types, richness of species or habitat-types, the presence of imperiled species, umbrella species, species endemism, local abundances of particular species or habitat types, population viability, metrics of habitat quality, metrics of ecological integrity, or economic efficiency. These factors quantify different aspects of value, and hence, a truly comprehensive index would include all of them, however, the available data preclude accurate estimates for most of them. The challenge we faced was to develop an index of relative conservation value that respected the limitations imposed by the currently available spatial data but still served as a useful, credible indicator of relative value.

Even with perfect spatial data for species occurrences and highly reliable models for habitat quality assessing the relative value of places for wildlife habitats would remain challenging because measures of conservation "value" or "importance" are normative. There is no purely objective conservation "value" that can be empirically validated because conservation value is based on one's belief about what is valuable, and therefore, it is influenced by subjective personal values. How various data should be assembled into an indicator of value may be different for each person, and therefore, a multitude of different credible indicators can be devised. Nevertheless, scientists may reach consensus on what factors should be used to indicate value and on the relative influence of those factors.

Terrestrial Habitats Assessment

The terrestrial habitats assessment focused on the principle process that currently dictates the quantity and quality of habitats in the Puget Sound Basin – land use. Prior to European settlement, the most important landscape-scale terrestrial process for creating and maintaining habitats in the Puget Sound Basin was fire, but over the past century wildfire has been effectively eliminated from the Puget Sound lowlands and Cascades foothills. In the lower-elevation landscapes of the Basin, where city and county governments have jurisdiction over land use, the historical landscape-scale process no longer operates at a landscape scale. The dominant large-scale disturbances are now those related to human land use which has created spatial gradients in landscape integrity.

The main challenge we faced in the terrestrial habitats assessments was the limitations imposed by the currently available spatial data. To compensate for the lack of data we utilized a "coarse filter-fine filter" approach. Coarse-filter elements are usually habitat types; the theory being that conserving habitat types will also conserve the vast majority of species associated with those habitat types. Fine-filter elements are usually those rare or imperiled species we believe will not be conserved by conserving habitat types alone. Our coarse filter-fine filter approach was somewhat unconventional because the principle coarse filters were not habitat types; the coarse filter was landscape integrity within forest zones. Our fine filter elements were certain priority species and habitat types.

Relative conservation value was calculated in three stages. First, open-space blocks were identified. An open-space block is a contiguous area containing land uses – such as commercial forest, agriculture, parks, and designated open-space – that maintain natural or quasi-natural vegetation which serve as habitats for native wildlife. Second, landscape integrity of the open-space blocks was assessed. Landscape integrity is the degree to which a landscape can support and maintain a biological community that has species composition, diversity, and functional organization comparable to those of natural landscapes in a region. In the third stage landscape integrity of open-space blocks was combined with spatial data for priority species and oak-grassland habitat types. The resulting product was an index of relative conservation value for assessment units (AUs), which were small watersheds with an average size of about 5 square miles.

The results of the terrestrial habitats assessment were not surprising. Relative conservation value exhibits an obvious gradient with a minimum in the Seattle-Tacoma metropolis and a maximum in the large blocks of undeveloped forest land that begin in the foothills of the Cascades or Olympic Mountains. In other words, at the landscape scale, relative conservation follows the classic urban-to-wildland gradient. In Pierce, King, and Snohomish counties that gradient runs roughly west-to-east. The exceptions to this gradient pattern are: 1) mouths of major rivers that are relatively undeveloped, such as the Nisqually, Skagit, and Nooksack, which support large concentrations of waterfowl and shorebirds; and 2) the oak-grassland habitat types in and around Fort Lewis.

The results depict a mixed suburban-rural “decision space” sandwiched between the large urban areas on Puget Sound and the relatively undeveloped foothills. Over the coming decades each county’s comprehensive plan will largely determine how much, where, and what kind of development will occur within that decision space.

Freshwater Lotic Habitats Assessment

Our freshwater habitats assessment focuses on the dominant property of lotic systems − connectivity. A watershed is comprised of a network of connected channels that funnel materials – predominantly, water, sediment, and wood − from the watershed’s headwaters down to its mouth. As materials move downhill through the network they provide both the matter and energy for the processes that build, destroy, and rebuild aquatic habitats. Habitat quality in stream reaches is determined by both local processes, such as hillslope runoff, bank erosion, channel scouring, and wood recruitment, and the same processes occurring remotely upstream. The quality of habitats in a stream reach is affected by conditions occurring upstream, and the conditions of that same reach affect habitat quality downstream. Therefore, our assessment of relative conservation value entails both an assessment of conditions upstream of each AU and an assessment of habitats downstream of each AU.

Over the past decade the dominant conservation issue for in the Puget Sound Basin has been salmon. Three salmon species in Puget Sound – Chinook, steelhead, and summer chum in Hood Canal − are currently listed as threatened under the federal Endangered Species Act, and a fourth, coho, is a candidate for listing. Consequently, the conservation of salmon and their habitats has garnered considerable attention and has manifested a number of assessments and plans. Each of the Salmon Recovery Lead Entities has done their own assessments to support their recovery plans for chinook and steelhead. The work done by lead entities serves a particular purpose, is highly attuned to local knowledge, and has involved local stakeholders. Therefore, our assessment is not a substitute for the assessments and plans of the lead entities.

While in-depth WRIA-level assessments have been done by salmon recovery lead entities, salmon still play a central in our assessment for several reasons. First, most lead entities only addressed listed species: Chinook, steelhead, and in some WRIAs, chum and bull trout. Second, salmonid species are the dominant vertebrate species in the lotic systems of Puget Sound Basin. Third, the eight salmonid species included in our assessment collectively inhabit an extensive geographic range in Puget Sound Basin. Fourth, occurrence data for salmonids constitute the most comprehensive and accurate data for any species inhabiting lotic systems. The last three reasons led us to utilize salmonids as umbrella species, i.e, species whose conservation confer a protective umbrella to numerous other co-occurring species.

The AUs were the same as those in the terrestrial habitats assessment, and for each we calculated an index of relative conservation value. The index had three components: the density of hydro-geomorphic features, local salmonid habitats, and the accumulative downstream habitats. That is, the relative value of a small watershed is based on: (1) the density of wetlands and undeveloped floodplains inside it, (2) the quantity and quality of salmonid habitats inside it, and (3) the quantity and quality of salmonid habitats outside and downstream of it. Quantity and quality of habitats were assessed for eight salmonid species.

We examined relative conservation value from two perspectives that reflect a quantity versus quality dichotomy. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's single most significant contribution, i.e., the quality a place contributes. The two perspectives result in different rankings of AUs. Together these two perspectives revealed that only 4% of AUs obtained high relative scores for all three index components but about 50% of AUs obtained high relative scores for at least 1 component. An additional 25% of AUs obtained moderate scores for at least one component. Hence, seventy-five percent of AUs in a typical WRIA obtain moderate to high relative scores for at least one component of the index because in a typical WRIA salmonid habitats are ubiquitous and salmonid habitats are physically connected to upstream conditions. These facts highlight the potential difficulties of future land use planning that strives to conserve freshwater lotic habitats.

Marine Shoreline Habitats Assessment

In contrast to the terrestrial and freshwater assessments, the marine shoreline assessment had comprehensive, reasonably accurate occurrence data for variety of animals and plants. Given the quality of data for the shorelines of Puget Sound, we believed an assessment based on the presence and density of species would provide a credible indicator of relative conservation value. The overarching assumption of our approach was that the relative value of shorelines for the conservation of fish and wildlife habitats is mostly a function of the presence and density of species for which we collect occurrence data. In general, we collect occurrence data for certain species because 1) humans harvest those species, 2) we are concerned about the status of those species (e.g. threatened or endangered species), or 3) we are concerned about the management of those species (e.g., species highly sensitive to human disturbances). In other words, we collect data on those species and habitats we care most about. Therefore, an assessment based on these data should indicate those places we should care most about for the conservation of fish and wildlife habitats.

The assessment units for marine shoreline habitats were small shoreline reaches with an average length of 0.24 miles, and for each we calculated an index of relative conservation value. The index had 41 components which included eight shellfish[1] species or species groups of commercial/recreational interest, urchins, three forage fish species, eight salmonid species, numerous bird species, pinnipeds, kelp, eelgrass, surfgrass, and wetlands.

Like the freshwater habitats assessment, we examined relative conservation value from two perspectives that reflect a quantity versus quality dichotomy. Also like the freshwater habitats assessment, these two perspectives showed that while very few shorelines scored high for all components, a over half of all shorelines scored moderate to high for at least one component. This has serious implications under Washington's Shoreline Management Act (RCW 90.58). The governing principles (WAC 173-26-186) of the shoreline guidelines (WAC 173-26-176) established under the SMA

[1] Shellfish includes both mollusks, such as butter clam, and crustaceans, such as Dungeness crab.

state, “Local [shoreline] master programs shall include policies and regulations designed to achieve no net loss of those ecological functions.” Any shoreline segment with composite index score greater than zero contains or is in close proximity to at least one ecological function, namely, a habitat function. Local jurisdictions must address the protection of habitat functions, and as the data show, habitat functions occur nearly everywhere along the shoreline of Puget Sound. However, the type and degree of protection required for each habitat function will vary greatly.

While our assessment shows that nearly all marine shorelines contains or are in close proximity to at least one habitat function, our assessment does not account for all habitat functions. For instance, we lacked occurrence data for the nearshore rearing habitats of juvenile salmonids and the rearing habitats of juvenile Dungeness crab. These are essential habitats functions that support vital commercial and recreational fisheries. Investments in data collection are needed to map the locations and quality of these and other habitat functions.

The results of our marine shoreline assessment should be used in conjunction with the assessments done by the Puget Sound Nearshore Ecosystem Restoration Project (PSNERP). Our index of relative conservation value is based on habitat functions. PSNERP’s assessments emphasize ecosystem processes and structures. Habitat functions are dependent upon ecosystem processes and structures. Therefore, integration of these complementary assessments will provide a more comprehensive understanding with which to make management decisions affecting nearshore ecosystems.

Using the Terrestrial, Freshwater, and Marine Shoreline Assessment

The main application of these assessments is to guide local land use zoning that occurs at the scale of 100s to 1000s of acres. County governments should use the results of the terrestrial and freshwater assessments to direct expansion of urban growth areas or new residential development to places that will minimally impact fish and wildlife habitats. The first areas to develop or develop more densely are those places (i.e., AUs) at the lowest end of relative conservation value. New development should be avoided in places at the highest end of conservation value. When directing new development toward lower value places, local jurisdictions should institute policies and regulations that protect the functions and values of critical areas.

The results of the marine shoreline habitats assessment should be used to guide the designation of shoreline land use zones that will achieve no net loss of the habitat functions that currently exist along shorelines. The marine shoreline assessment can also help prioritize shoreline restoration within oceanographic sub-basins.

The results of our terrestrial, freshwater, and marine shoreline habitat assessments constitute a subset of information that should be considered by county and city governments for land use planning. Our assessments should be complemented with other information such as that provided by Salmon Recovery Lead Entities, the Puget Sound Nearshore Ecosystem Restoration Project, or local biological surveys. Assembling, organizing, and integrating scientific information from diverse sources is a continual challenge for local governments. In volume 3 of the watershed characterization project, we will provide guidance to effect this integration. To assist local governments overcome this challenge, WDFW and Ecology have formed a watershed characterization technical assistance team (WCTAT) consisting of state agency scientists with expertise in wildlife biology, fish biology, wetlands, hydrology, geomorphology, and modeling.

Caveats

When using the habitat assessments keep in mind the following limitations. First, our assessments are landscape-scale assessments, and consequently, do not address habitat issues that are best addressed through finer-scale studies. Finer-scale or site-level actions, such as critical area ordinances that protect nest sites, riparian areas, or wetlands, will remain essential to the success of local habitat conservation efforts. We did our assessments with the expectation that finer-scale studies will be done by local governments as the need arises. When developing land use plans, city and county governments should evaluate the need for finer-scale information and collect it where needed.

Second, our indices of relative conservation value are not comprehensive. The assessments are not comprehensive in three respects. First, the assessments did not explicitly include all species because we lack reasonably accurate occurrence data or habitat models for most species, even for most priority species such as Keen's myotis, pileated woodpecker, band-tailed pigeon, western toad, and Pacific lamprey. Second, the assessments did not fully address habitat connectivity because it has been or will be addressed through other assessments, as explained below. Third, we narrowed the spatial extent of the terrestrial and freshwater assessments to areas that fall under the jurisdiction of city and county land use plans. That is, we did not address species or habitats that are mostly confined to higher elevations (>2000 ft) on public lands.

In the terrestrial and freshwater assessments, however, we did implicitly address nearly all species. In the terrestrial assessment, relative conservation value was mainly a function of landscape integrity. The presence of PHS habitats was also an important factor but because of their relatively small spatial extent, PHS habitats were much less influential on relative conservation value than landscape integrity. Basing conservation value on landscape integrity was essentially a coarse-filter approach which assumes that areas with high landscape integrity will provide high quality habitat for the majority of wildlife species. In the freshwater assessment, conservation value was based largely on the quantity and quality of salmonid habitats. This was effectively an umbrella-species approach which assumes that areas protected for salmonid habitats will also protect habitats for the majority of other species in lotic habitats.

In the marine shoreline assessment, conservation value was calculated as a composite index consisting of 41 diverse components. This was effectively a richness approach which assumes that our 41 components can serve as adequate surrogates for the majority of species in marine shoreline habitats of Puget Sound. However, the available occurrence data were biased towards harvested species. Therefore, the marine assessment might be better characterized as an ecosystem services approach in which habitats are ecosystem functions that support the ecosystem service of food provision.

Because of the assumptions and simplifications we made, the terrestrial, freshwater, and marine shoreline assessments may not adequately address the particular habitat needs of rare or imperiled (i.e., state or federally listed) species or species highly susceptible to human disturbance. If rare or imperiled species inhabit a local jurisdiction, then the special needs of such species should be specifically addressed in local land use plans.

Third, one particularly important aspect of biodiversity conservation which we did not adequately address was connectivity. Landscape integrity in the terrestrial assessment incorporated factors which address habitat connectivity but only obliquely. More detailed assessments on habitat connectivity may be necessary. Connections between habitat patches can be provided by smaller-scale features such as riparian corridors that are best delineated through finer-scale assessments. State-wide connectivity has been addressed by the Washington Wildlife Habitat Connectivity Working Group (WHCWG 2010). The results of the WHCWG assessment should be incorporated into regional land use planning. Upstream and downstream relationships within the stream network were foundational to the freshwater assessment, however, that assessment did not explicitly include artificial barriers to fish passage. The freshwater assessment addressed stream connectivity indirectly through occurrence data that documented the presence of andromous salmonids and expert judgments about which streams could support anadromous fish when artificial barriers are removed.

Along marine shorelines the main connectivity issue is the movement of sediments within littoral drift cells. Maintaining process connectivity within drift cells has been the focus of PSNERP. Integrating our marine shoreline assessment with PSNERPs assessments of drift cells will help local governments prioritize shorelines for protection and restoration of connectivity.

Fourth, there is no purely objective “conservation value” that can be empirically validated. “Value” is based on one’s belief about what is valuable, and therefore, subjective. Furthermore, there is a wide variety of potential credible models of conservation value that could have been constructed for this assessment. Our models of habitat conservation value for the three assessments were based on a number of subjective judgments for which there was uncertainty: which factors to include, their relative influence, and how to assemble them. Through numerous meetings with experts and intensive peer review we believe we have developed useful, scientifically credible indices of relative conservation value.

Lastly, as data, technology, and knowledge improves over time better assessments will emerge. Other initiatives will reassess habitats in the Puget Sound Basin. For Instance, the Western Governors Association has initiated a project to identify “crucial habitats” throughout the western states (WGWC 2011). WDFW is participating in that initiative, and the results of the western governors’ assessment could supplement or supplant our terrestrial or freshwater assessments.

Part 1:
Introduction and Background

1.1 Introduction

Over the next two decades over 1 million additional people are expected to inhabit the Puget Sound Basin (OFM 2007). Thousands of acres of agricultural and timber lands will be converted to residential and commercial uses in order to accommodate this phenomenal growth. In addition to providing valuable commodities these “working lands”, both agricultural and timber lands, provide habitats for wildlife. To ensure the health and well-being of their citizens, promote orderly and efficient land use, and protect natural resources, city and county governments implement comprehensive plans and regulatory land use zoning. Natural resources include fish and wildlife. In general, conversion of agricultural and timber lands to residential or commercial uses adversely impacts landscape integrity and the composition of native biotic communities (Hansen et al. 2005, Azerrad et al. 2009). Effective land use zoning can result in “smart” growth (Daniels 2001, Alexander and Tomalty 2002) that minimizes the loss and degradation of fish and wildlife habitats. Furthermore, city and county governments are instituting innovative transfer of development rights programs (TDRs) which also establish zones − receiving and sending areas. Receiving areas are places already impacted by development where new development should be concentrated and sending areas are places with very little development where working lands and natural resources should be conserved. To fully realize smart growth, comprehensive land use plans must be based on scientifically-credible information that indicates the most important places for the conservation of fish and wildlife habitats – places where development should be avoided. Our purpose is to provide useful, scientifically-credible information for smarter growth in the Puget Sound Basin. Our task is to assess the relative value of places throughout the Basin for the conservation of fish and wildlife habitats.

The Puget Sound Watershed Characterization Project is a set of spatially explicit assessments that provide information for regional, county, and watershed-based planning. It is a coarse-scale decision-support tool that should lead to better decisions regarding land use and more effective conservation of the region’s natural resources. The assessments cover water resources − both water flow and water quality − and fish and wildlife habitats – in terrestrial, freshwater, and marine shoreline environments – within the entire Puget Sound Basin. The Department of Ecology is leading the assessments for water resources and the Department of Fish and Wildlife is leading the assessments for habitats. This volume describes the terrestrial, freshwater, and marine shoreline habitats assessments. Refer to Stanley et al. (2011) for descriptions of the water flow and water quality assessments.

The assessments provide a regional perspective on the *relative* value of small watersheds (average size 4.7 square miles) and marine shorelines for the conservation of water resources and habitats. The assessments’ primary products are maps that show the relative value of these small watersheds (hereafter referred to as assessment units or AUs) or marine shorelines. Their relative values are expressed through quantitative indices that can be used to rank AUs within a county or a water resources inventory area (WRIA). The indices and the data used to calculate the indices are stored in a geographic database. The targeted users of the assessments are land use planners of city or county governments.

Because of differences in spatial dimensions, spatial scale, data quality, and ecosystem-level processes the fish and wildlife assessment was broken into three separate assessments: terrestrial, freshwater, and marine shoreline. Parts 2, 3, and 4 of this report describe the terrestrial, freshwater, and marine shoreline assessments, respectively.

1.1.1. Relationship to Other Assessments

The Puget Sound Watershed Characterization Project is one of many spatially-explicit assessments covering the Puget Sound Basin that have identified, prioritized, ranked, or scored places based on their conservation value. Every one of these assessments served a unique purpose and was conducted in a unique way. For example, prior to the Puget Sound Watershed Characterization Project, WDFW and The Nature Conservancy (TNC) completed nine ecoregional assessments covering all of Washington State. Four of these ecoregional assessments overlap parts of the Puget Sound Basin (Floberg et al. 2004, Iachetti et al. 2006, Vander Schaaf et al. 2006, Popper et al. 2007), but they aren't appropriate for local land use planning because: 1) the assessments' main purpose was to identify an efficient, fixed set of places to be targeted for conservation action; and 2) the ecoregions were assessed as independent entities, and consequently, their separate results are incompatible and cannot be merged into a single unified assessment for the Basin. Furthermore, the ecoregional assessments did not address water resources. The Watershed Characterization Project provides city and county governments with consistent, comprehensive information about water and habitat resources across their entire jurisdiction.

Another Assessment for Lotic Freshwater Habitats?

Over the past decade the dominant conservation issue for in the Puget Sound Basin has been salmon. Three salmon species in Puget Sound – Chinook, steelhead, and summer chum in Hood Canal – are currently listed as threatened under the federal Endangered Species Act, and a fourth, coho, is a candidate for listing[2]. Consequently, the conservation of salmon and their habitats has garnered considerable attention and has manifested a number of assessments or plans. For instance, The Trust for Public Land (TPL) and TNC did their own assessments in 2000 and 2006, respectively (Frissell et al. 2000, Skidmore 2006). TPL's assessment identified important places for the conservation of salmon and TNC's identified important places for all freshwater biodiversity. These two assessments served specific purposes for particular clients, and therefore, may not meet the needs of city or county governments engaged in local land use planning.

Each of the Salmon Recovery Lead Entities has done their own assessments to support their recovery plans for Chinook and steelhead (e.g., East Kitsap 2004, Pierce County 2004, Snohomish County 2005). The work done by lead entities serves a particular purpose, is highly attuned to local knowledge, and has involved local stakeholders, and therefore, our assessment is not a substitute for the assessments and plans of the lead entities. Furthermore, the lead entities' assessments and plans generally focused on habitat restoration; our habitat assessments do not address the relative value of places for restoration.

Our freshwater assessment is needed for several reasons. First, the assessments done by Salmon Recovery Lead Entities focused on listed species: Chinook, steelhead, and in some WRIAs, chum or bull trout. Our assessment incorporates all salmonids. Second, the lead entities mainly focused on identifying sites for habitat restoration projects, and nearly all restoration sites were in or adjacent to water bodies. Most assessments done by the lead entities' neglected to provide landscape-level information such as identifying areas in their watersheds where future residential or commercial development would have the greatest adverse impacts on salmonid habitats. Our assessment does that. Third, the Department of Ecology has completed Basin-wide assessments for water resources in which they assigned values of relative importance to AUs. Integrating the water resources assessments

[2] For the purposes of the ESA, the U.S. Fish and Wildlife Service divides salmonid species in evolutionary significant units (ESUs) which are conceptually similar to subspecies. The Puget Sound ESUs of Chinook and steelhead are listed as threatened.

with the fish and wildlife habitat assessments will create a multi-resource depiction of the most important AUs in the Basin. The terrestrial and freshwater habitat assessments utilize the same spatial units created for the water flow and water quality assessments in order to facilitate a seamless integration of all four assessments. Finally, the assessment methods employed by lead entities were different for every lead entity. Therefore, the results of the lead entity assessments are not comparable. Our assessment method is applied uniformly across the Basin, and thus, provides a consistent regional perspective.

The Puget Sound Nearshore Ecosystem Restoration Project

The Puget Sound Nearshore Ecosystem Restoration Project (PSNERP) was formally initiated as a general investigation feasibility study in September 2001, through a cost-share agreement between the U.S. Army Corps of Engineers and the State of Washington, represented by the Washington Department of Fish and Wildlife (WDFW). PSNERP has completed a feasibility study to assess ecosystem degradation in the Puget Sound Basin; to formulate, evaluate, and screen potential solutions to ecosystem degradation; and to recommend a series of actions (Cereghino et al. 2012).

The habitat assessments conducted by WDFW for the Puget Sound Watershed Characterization Project were initiated in 2010. In 2011 PSNERP and the authors of this report recognized that each group was assessing complementary aspects of nearshore ecosystems. Our index of relative conservation value is based on habitat functions. PSNERP's assessments emphasize ecosystem processes and structures. Habitat functions are dependent upon properties of ecosystem processes and structures. Therefore, integration of these complementary assessments will provide a more comprehensive understanding with which to make management decisions affecting the nearshore ecosystems. The results of our marine shoreline assessment should be used in conjunction with the assessments done by the Puget Sound Nearshore Ecosystem Restoration Project (PSNERP).

1.2 Philosophical and Methodological Foundations

Our task was to assess the relative value of places for the conservation of fish and wildlife habitats. Our approach for assessing relative value was the calculation of indices. An index reduces a complex, multi-dimensional system down to a single number. The resulting simplification facilitates planning and policy making. The Dow Jones Industrial Average, for example, is a stock market index that tracks the day-to-day progress of a highly complex economic system with a single number that is recalculated each business day. The Dow Jones is intended to provide a big-picture view of the industrial sector of the economy over time. The Dow Jones cannot be used to judge the performance of any particular industry. To gain a better understanding of how various industries have performed, one must examine the many components of the Dow Jones or refer to other industry-specific information. Likewise, our indices provide a big-picture view of relative conservation value over the landscape within an entire county or water resource inventory area (WRIA). Summary indices such as ours cannot be used to understand the status of particular species or habitats or to design site-level projects. Our indices may even mask some important aspects of conservation value, but when interpreted properly, the indices can facilitate better decisions regarding habitat protection and land use (Failing and Gregory 2003).

Certain places in a region are readily identified as valuable or even irreplaceable because they contain rare habitat-types, imperiled species, or abundant wildlife. For instance, in the Puget Trough Ecoregion, the prairies on Fort Lewis, the tidelands at the Nisqually River delta, the waterfowl over-wintering areas of the Skagit River delta, Protection Island with its dense colonies of breeding birds, and the Elwha River are universally recognized by biologists as crucial places for habitat conservation. The value of such places is obvious and absolute – experts are certain that these places should be protected or restored for their ecological values. Most other places lack rare habitats, imperiled species, or abundant wildlife. Such places may have value for the conservation of fish and wildlife habitats, but they lack those qualities that would make their protection indisputable. The value of places with “common” habitats can be assessed but only in a *relative* sense, and decisions regarding their protection must be based on *relative* value. Hence, for the multitude of places that contain only common species or common habitats, our assessment cannot determine whether site A or site B should be protected. Our assessment can only determine that site A is relatively more or less valuable for wildlife habitat than site B, and therefore, site A should be a higher or lower priority for habitat protection than site B.

The relative value of a place for the conservation of fish and wildlife habitats can be based on a variety of different factors: the presence of rare species or habitat-types (Prendergast et al. 1993, Kerr 1997), richness of species or habitat-types (Williams et al. 1996), the presence of imperiled species (i.e., listed as threatened or endangered), habitat for umbrella species (Poiani et al. 2001), species endemism (Orme et al. 2005), local abundances of particular species or habitat types (Winston and Angermeier 1995, Pearce and Ferrier 2001), metrics of habitat quality (Root et al. 2003), metrics of ecological integrity (Andreasen et al. 2001), or through optimization algorithms (Wilhere et al. 2008). These factors quantify different aspects of value, and hence, a truly comprehensive assessment would include all of them, however, the available data preclude accurate estimates for most of them.

1.2.1. The Challenges of Assessing Relative Conservation Value

Empirical data on the locations of wildlife species collected by WDFW and other agencies generally focus on imperiled species or harvested species. For the vast majority of other wildlife species, site-scale location data are based on incidental observations, incomplete surveys, or are out of date. Furthermore, data accuracy tends to be a function of a species’ sightability – location data for highly visible species (e.g., large bodied in open habitats) tends to be more accurate than data for hard to see species (e.g., small bodied with cryptic markings in densely vegetated habitats). For nearly all vertebrate species, comprehensive data on fish and wildlife locations are available as range maps (e.g., Johnson and Cassidy 1997, Wydoski and Whitney 2003) but these can be highly inaccurate at spatial resolutions of about 4 square miles or more. For the locations of habitat types, satisfactory empirical data are available for rare or imperiled habitats types, such as oak woodlands and prairies. Location data for other habitat types is available as land cover maps derived from satellite remotely-sensed images. These land cover data tend to have either low classification accuracy (e.g., 35 % error for ecological systems in western Washington [Sanborn 2007]) or low thematic precision, i.e., a small number of land cover categories. Both shortcomings preclude an accurate mapping of habitat types, species-specific habitats, or habitat quality.

Even with perfect spatial data for species occurrences and highly reliable models for habitat quality assessing the relative value of places for wildlife habitats would remain challenging because measures of "value" or "importance" are normative (Turnhout et al. 2007). There is no purely objective "value" that can be empirically validated because conservation value is based on one's belief about what is valuable, and therefore, it is influenced by subjective personal values. For example, people will answer the following question differently (Figure 1.1): what is most important, a place rich with common species, a place with a few rare species, or a place with commercially valuable species? Likewise, how various data should be assembled into an indicator of value may be different for each person, and therefore, a multitude of different credible indicators can be devised. Nevertheless, scientists may reach consensus on what factors should be used to indicate value and on the relative influence of those factors.

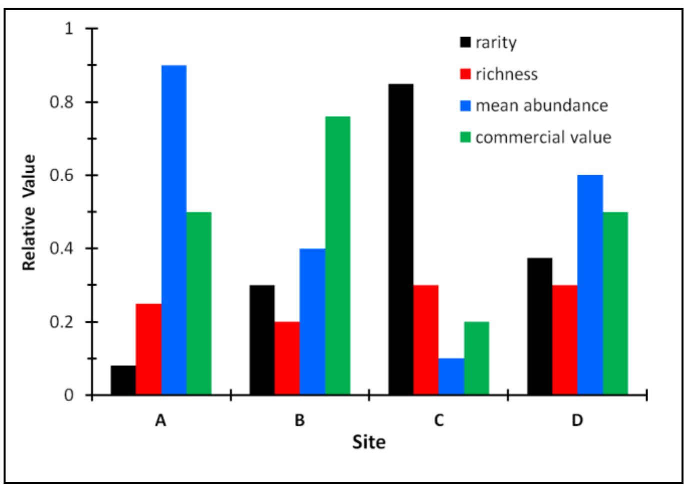


Figure 1.1. The normative challenge of determining relative value of places for fish and wildlife habitat conservation. Bars show results of hypothetical assessments for sites A, B, C, and D. All sites have habitat value, but which site is most valuable? If protection of rare species is a priority then site C. If protecting commercially valuable species is a priority then site B. If all factors are considered equally then D is the most valuable.

In summary, practical measures of conservation value are constrained by the types, quantity, and quality of available data. Furthermore, "conservation value" is not descriptive, it is normative, and hence, transparency regarding normative judgments is essential. The challenge we faced was to develop an assessment that respected the limitations imposed by the currently available spatial data but still served as a useful, credible indicator of relative conservation value.

1.2.1. Designing an Index of Relative Conservation Value

An index of places' relative conservation value for fish and wildlife habitats must necessarily be subjective. In effect, the index is a model of our beliefs about what makes different places relatively more or less valuable for habitat conservation. Various data can be assembled multiple different ways to yield a variety of potential models (Niemeijer 2002). The range of potential models will be narrowed by the types, quantity, and quality of available data and by the intended application of the index (Figure 1.2). Some potential models can be eliminated because we lack the data or data of sufficient quality needed to implement them. Understanding the scope and spatial scale of the intended application should lead to models with the minimum scope at the proper scale. An assessment should cover the narrowest scope that still meets a client's needs because as scope increases both complexity and ecological variation increase. Increasing complexity increases epistemic uncertainty, which arises from our lack of knowledge, and increasing ecological variation increases aleatory uncertainty, which is due to the intrinsic randomness of natural systems. In short, as the scope of an assessment increases our confidence in the results often decreases.

To simplify the habitat assessments and enhance our confidence in their results we narrowed the scope of the assessments to private lands. Our principal clients, i.e., local governments, want an assessment that informs land use planning. Nearly all land use planning done by local governments affect only private lands. Private lands comprise about 50% of the land area in Puget Sound Basin. An assessment that encompassed both private and public lands would be more complex than an assessment that covers only private lands because ecological systems, land use, management practices, and the consequent habitats on public lands tend to be quite different than those on private lands. Narrowing the scope to private lands facilitates development of simpler models having less uncertainty because focusing on private lands allowed us to ignore the habitat types and wildlife species that reside predominantly on public lands (e.g., mountain hemlock forest, mountain goat, Olympic marmots), and therefore, not significantly affected by local land use decisions. However, the influence of public lands on the habitat quality of private lands was integrated into our terrestrial and freshwater assessments. For instance, in our assessment, private forest adjacent to large blocks of public forests have greater landscape integrity than identical private forests distant from large blocks of public forest.

A model's spatial scale should match the scale of decision making. Spatial scale has two components extent and grain. Extent is the entire area affected by a decision. Grain is the average size of places or sites for which decisions are being made. Obviously, the extent of an assessment must not be smaller than the extent affected by the decision making process, but neither should the extent be much larger. As the spatial extent of an assessment increases ecological variation may also increase, and hence, uncertainty. Assessments done at a coarser grain than decision making may have little value to decision makers. For instance, if decisions are to be made about 10-acre parcels, then an assessment based on 5,000-acre watersheds is of little use. An assessment done with too fine a grain can often be translated to a coarser grain but there are potential problems with many finer-grained data. For instance, higher resolution land cover data sometimes have unknown or low classification accuracy (e.g., 35 % error for ecological systems in western Washington [Sanborn 2007]). The spatial grain for the terrestrial habitats assessment is the AUs, which are small catchments roughly 1 to 10 square miles in size (640 to 6400 acres). Land use zoning by county governments creates zones on the order of 100 to 10,000 acres, and hence, the spatial grain of the habitat assessment roughly matches the scale of zoning. County governments make many site-level decisions but the scale of our AUs does not match that smaller spatial scale.

After the scope and scale of the assessment have been well described, there will still be a wide variety of potential models. The number of potential models will be reduced by the availability and quality of the data needed for each model, but there may still be a wide variety of candidate models. A subset of reasonable candidate models can be further reduced through the principle of parsimony. For our purposes, parsimony means that the number of factors incorporated into the model should be the smallest number that adequately represents the conservation value of a place. In some respects selecting factors to incorporate into our model is similar to selecting independent variables in a regression model. The factors should (1) have a significant influence on relative conservation value, and (2) have low correlation with other factors included in the model.

The structure of a model, any type of model, is ultimately based on subjective judgment; however, unlike a regression model, we had no established statistical methodology with which to guide our construction of the best model. We could not test which factors (i.e., independent variables) have a "significant" influence on relative conservation value or estimate multi-collinearity amongst factors. Hence, the selection of factors was based on best professional judgment. Our model of relative conservation value was based on a number of subjective judgments for which there was uncertainty: which factors to include, the relative influence each factor should have on conservation value, and how to assemble the factors.

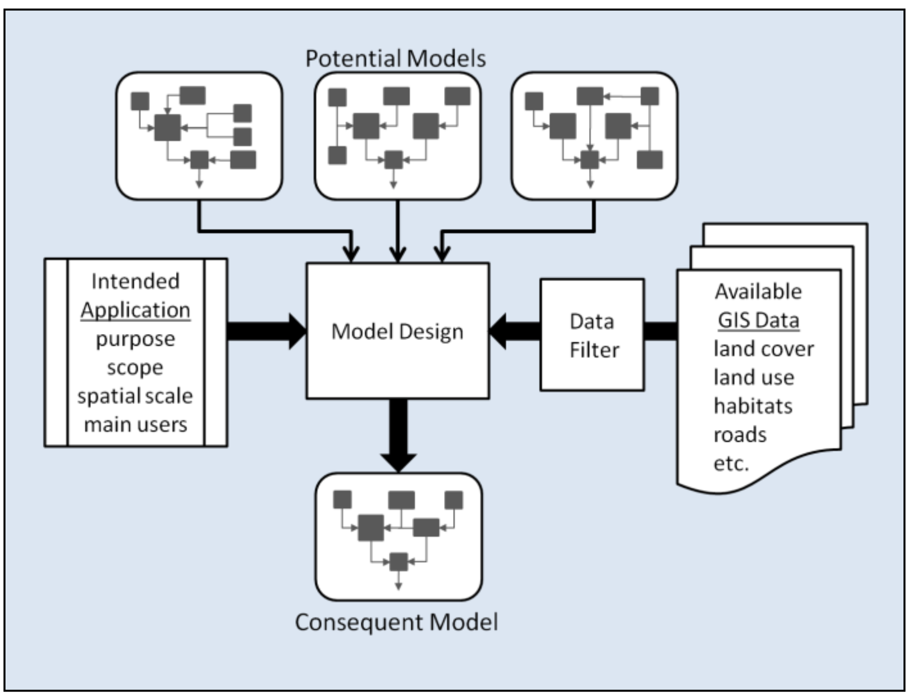


Figure 1.2. Conceptual model for the design of our fish and wildlife habitat assessments. There are many ways to assess conservation "value" or "importance" of places for fish and wildlife habitat conservation. However, the data required to implement most models are not available or of unacceptable quality, and hence, model design involves a number of assumptions and compromises. Models can be simplified by considering the model's intended application.

1.2.2. Good Modeling Practice

According to Schmolke et al. (2010) elements of good modeling practice include conceptual models, quantifying uncertainty, sensitivity analysis, verification, validation, and peer review.

Conceptual Models

A conceptual model is a simplified representation of a complex system that emphasizes the interrelationships among the system's major elements. A conceptual model states the major assumptions and limitations of our current understanding of the system. The conceptual model is the basis for the components, structure, and operation of the quantitative model. Each of our three assessments begins with a conceptual model.

Uncertainty

A favorite saying amongst statisticians is "all models are wrong, but some are useful."[3] This tongue-in-cheek exaggeration conveys a caveat about models. What statisticians really mean is all models have statistical error, also known as uncertainty. Even models constructed by the best scientists using the best data have uncertainty, i.e., the model's predictions are estimates that are never exactly correct. Nevertheless, if a model's predictions are close enough to correct most of the time, then we may have a useful model. Our assessments should be used with this same caveat in mind – the results of our assessments are uncertain.

Depictions of uncertainty indicate how confident we can be about relative conservation value. For instance, uncertainty analysis (explained below) may show that even though two AUs have different scores for relative conservation value, we cannot be certain that their scores are truly different (Figure 1.3). The uncertainty may be so large in some cases that managers should treat the two AUs as if they have equal scores.

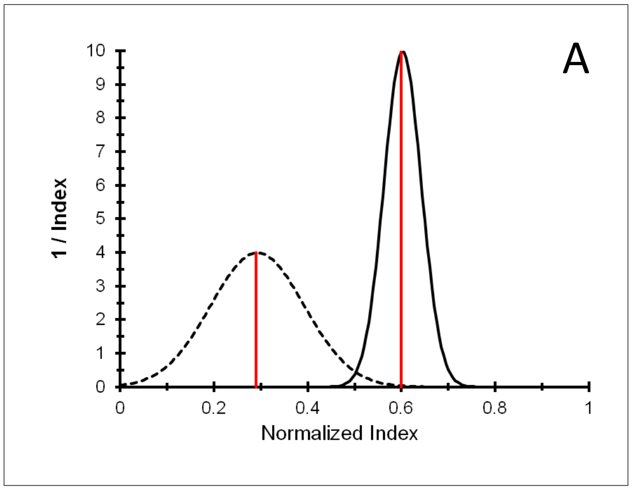
We can make generalizations about when we can be more confident about scores and when we should be more wary. For assessments of this type, relative values at the extreme ends of the range (e.g., the top and bottom deciles) tend to have the smallest uncertainty, and places with more moderate values (5^{th} and 6^{th} deciles, for instance) tend to have larger uncertainty. Hence, a set of AUs with different values but in the moderate range may have effectively the same value. Greater uncertainty does not mean that an AU has lesser value than that estimated through the assessment. Greater uncertainty means that the actual relative conservation value could be larger or smaller than the estimated value.

Uncertainty analysis establishes the confidence limits of a model and evaluates the robustness of a model's output to input or parameter uncertainties. If a model is robust, then uncertainties may be attenuated at the output. That is, even with uncertainties about model inputs and parameters we may still have confidence in the model.

Our uncertainty analyses were done via Monte Carlo methods. We assigned a uniform or triangular probability distribution to each parameter in an index's equations. From these distributions parameter values were randomly selected and the index was recalculated for each AU – a process that was repeated thousands of time. The index value of each AU is effectively a separate model output, and

[3] This is paraphrased from a statement by the statistician George E. P. Box (Box and Draper 1987): "All models are wrong; the practical question is how wrong do they have to be to not be useful."

hence each AU has its own distribution of index values. The uncertainty associated with each AUs index value was calculated and displayed as a 90% confidence interval.



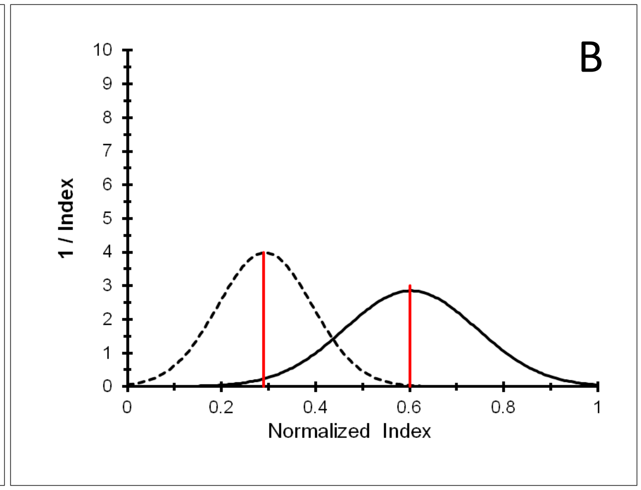


Figure 1.3. Hypothetical probability distributions representing uncertainty in calculation of relative conservation value index. Red lines are expected values. In both panels the two sites have expected values for the index of 0.29 and 0.6. If distributions for two sites overlap very little (panel A), then we can be confident that the two sites are different. If probability distributions have substantial overlap, then we consider the two sites to be not significantly different (panel B).

There is no standard methodology for conducting uncertainty analyses for indices such as ours. Our uncertainty analyses only addressed the potential uncertainties inherent to the subjective judgments made for parameter values. The parameters' distributions were based on expert judgment and spanned the range of reasonable values for each parameter. We did not examine other uncertainties that affect the model output. We did not address, for example, the uncertainty (i.e., errors) in spatial data layers such as land cover or the WDFW's wildlife occurrence data, nor did we address uncertainties regarding model structure. Addressing all uncertainties was entirely impractical. Therefore, our uncertainty analyses did not produce rigorous estimates of uncertainty. The uncertainty analyses serve as a reminder about the limitations of each assessment and a warning against making fine distinctions between AUs with approximately the same score.

Sensitivity Analysis

In addition to uncertain analysis, further evaluation of the indices was done with sensitivity analysis. Sensitivity analysis explores how much our conclusions would change if the model's inputs or parameter values were different. It reveals which model inputs or parameters are most influential on the model's output. If a model is highly sensitive to a parameter, then a small change in the parameter's value causes a large change in the model's output. The inputs or parameters to which the model is most sensitive are the ones to which we should pay the closest attention. For instance, if we want to improve the accuracy of a model but have limited resources, then we should focus data collection on those parameters the model is most sensitive to.

Our indices of relative conversation value that can be expressed as a function:

$$I = f(V_1, V_2, V_3, \ldots, V_n) \quad (1.1)$$

where the components V_j *equal* w_jX_j, X_j is a variable (or model input) representing a species or habitat, w_j is a weight (or parameter) that determines the relative influence of X_j, and n is the number of components that comprise I. The sensitivity of an index to a component, V_j, is defined as:

$$S_j = \Delta I / \Delta V_j \quad (1.2)$$

A normalized sensitivity, known as elasticity, is defined as:

$$E_j = (\Delta I / \Delta V_j)\,(V_j / I) \quad (1.3)$$

Elasticity is interpreted as percent change in the model output for a percent change in a component.

Sensitivity analysis was done by recalculating the index for all AUs with a single component altered by a small amount (+ 5%). The calculation was repeated for each component. The index value of each AU is effectively a separate model output, and hence, every AU has its own sensitivity to each component. Sensitivity is reported in two ways: 1) for each individual AU we calculated the mean sensitivity to all components, and 2) collectively for all AUs we calculated the mean sensitivity to each individual component. The former was calculated by averaging all the component sensitivities of an AU and the later was calculated by averaging all the AU sensitivities for each component.

Verification

Model verification checks that the model does what it's supposed to do (Rykiel 1996). Verification looks for errors in data processing and verifies that a model's calculations were done correctly. For complex models that process multiple large data sets, such as the models in our assessments, this is an essential task. We verified the indices by doing nearly all index calculations twice: once in Microsoft Excel (2007) and once in R (RDCT 2005). Spatial data analyses done in ArcGIS (version 10.0) were verified by having one analyst perform the task and a second analyst review the results.

Validation

Validation means that a model is demonstrated to be acceptable for its intended use (Rykiel 1996). The major intended use of our indices is to guide landscape-scale decisions regarding land use zoning. We evaluated the performance of the indices by comparing index scores against our knowledge of the Puget Sound Basin. We tested whether the index showed AUs we believed to be relatively more important as more important and showed AUs we believed to be relatively less important as less important. In other words, the validation test was based on the ranking of a subset of AUs.

Model validation often entails testing the accuracy of model predictions. For a statistical model, validation entails running independent data (i.e., data not used to develop the model) through the model to determine whether empirically measured responses fall within the model's prediction interval. Statistically rigorous model validation is objective. We could not do a rigorous validation because indices of conservation "value" or "importance" are subjective. There is no purely objective "importance" that can be empirically validated. "Importance" is based on one's belief of about what is important, and therefore, it is influenced by personal values.

Our model was not based on data but on professional expertise, which includes expert knowledge of ecological concepts, conservation principles, mathematics, modeling, and the relevant technical literature. Hence, our model can only be "validated" through peer-review by other scientists with the appropriate professional expertise.

Peer Review

A credible model is one in which managers or planners have sufficient confidence that they will use it for management or planning (Rykiel 1996). The credibility of our indices is built upon peer-review by other scientists with the appropriate professional expertise. We obtained peer-review in two ways. First, at two points during model development we convened a panel of experts to review and critique our models of relative conservation value. The critiques led to subsequent changes and improvements to the data and models. Second, we subjected our models to a more formal peer-review in which experts read our draft reports and submitted written comments. The written comments also led to changes and improvements to the models.

1.3 Overview of the Assessments' Methods

We did habitat assessments for three different environments: terrestrial, lotic freshwater, and marine shorelines. Because montane habitats are almost entirely located on public lands, the terrestrial assessment ignored montane habitats and concentrated on the Puget Trough lowlands and Cascade foothills. The freshwater assessment was restricted to lotic (i.e. flowing water) habitats because the major conservation issues facing lentic systems (i.e. ponds and lakes), such as intensive shoreline development, failing septic systems, introduction of non-native vertebrate species, and invasive species, are localized problems occurring at finer-scales than can be addressed by our assessment. The marine assessment was confined to shorelines because we lack data with which to assess the relative conservation value of deeper waters and the most direct impacts from development occur along shorelines.

The fish and wildlife assessment was broken into three separate assessments because of differences in spatial dimensions, spatial scale, data quality, and ecosystem-level processes. The marine shoreline is essentially a line 2470 miles long, lotic freshwater habitats are configured as a one-dimensional multi-branched network over 50,000 miles long, and terrestrial habitats are a two-dimensional surface covering 13,700 square miles (8.7 million acres). The differences in dimensions and scale manifest differences in data quality. Because the marine shoreline is one-dimensional and relatively short, the entire marine shoreline of Puget Sound has been surveyed for habitat types (Berry et al. 2001a) and is annually surveyed for birds (Nysewander et al. 2005). Consequently, the marine shorelines assessment had more accurate and comprehensive biological data than the terrestrial and freshwater assessments, and the approach adopted for the marine shoreline assessment exploited this higher quality data. Likewise, the approaches adopted for the terrestrial and freshwater assessments exploited the best spatial data available, however, for both assessments the best spatial data available imposed limitations on how we could assess relative conservation value.

This report covers three separate assessments for three different environments and also attempts to address two separate audiences: land use planners and scientists. Some planners or scientists may be interested in only one of the three assessments. In the interest of serving both audiences, some repetition could not be avoided. We apologize for any inconvenience.

1.3.1. Terrestrial Habitats Assessment

The terrestrial habitats assessment focused on the principle process that currently dictates the quantity and quality of habitats in the Puget Sound Basin – land use. Prior to European settlement, the most important landscape-scale terrestrial process for creating and maintaining habitats in the Puget Sound Basin was fire, but over the past century wildfire has been effectively eliminated from the Puget Sound lowlands and Cascades foothills. In the lower-elevation landscapes of the Basin, where city and county governments have jurisdiction over land use, the historical natural process no longer operates at a landscape scale. The dominant large-scale disturbances are now those related to human land use which have created spatial gradients in landscape integrity.

The main challenge we faced in the terrestrial habitats assessments was the limitations imposed by the currently available spatial data. To compensate for the lack of data we utilized a "coarse filter-fine filter" approach. Coarse-filter elements are usually habitat types; the theory being that conserving habitat types will also conserve the vast majority of species associated with those habitat types. Fine-filter elements are usually rare or imperiled species we believe will not be conserved by conserving habitat types alone. Our coarse filter-fine filter approach was somewhat unconventional because the principle coarse filters were not habitat types; the coarse filter was landscape integrity within forest zones. Our fine filter elements were certain priority species and habitat types.

Relative conservation value was calculated in three stages. In the first stage open-space blocks were identified and assessed for landscape integrity. An open-space block is a contiguous area containing land uses – such as commercial forest, agriculture, parks, and designated open-space – that maintain natural or quasi-natural vegetation which serve as habitats for native wildlife. In the second stage, the landscape integrity of open-space blocks was combined with PHS species data, including oak-grassland habitat types. Third stage consisted of calculating an index of relative conservation value for each assessment unit, which were small watersheds with an average size of about 5 square miles.

1.3.2. Freshwater Lotic Habitats Assessment

Our freshwater habitats assessment focuses on the dominant property of lotic systems − connectivity. A watershed is comprised of a network of connected channels that funnel materials – predominantly, water, sediment, and wood − from the watershed's headwaters down to its mouth. As materials move downhill through the network they provide both the matter and energy for the processes that build, destroy, and rebuild aquatic habitats. The habitat quality of stream reaches is determined by both local processes, such as hillslope runoff, bank erosion, channel scouring, and wood recruitment, and the same processes occurring remotely upstream. Aquatic habitat quality in a stream reach is affected by conditions occurring upstream, and the conditions of that same reach affect habitat quality downstream. Therefore, our assessment of relative conservation value entails both an assessment of conditions upstream and an assessment of habitats downstream.

While in-depth WRIA-level assessments have been done by salmon recovery lead entities, salmon still play a central in our assessment for several reasons. First, most lead entities only addressed listed species: Chinook, steelhead, and in some WRIAs, chum and bull trout. Second, salmonid species are the dominant vertebrate species in the lotic systems of Puget Sound Basin. Third, the eight salmonid species included in our assessment collectively inhabit an extensive geographic range in Puget Sound Basin. Fourth, occurrence data for salmonids constitute the most comprehensive and accurate data for any species inhabiting lotic systems. The last three reasons led us to utilize salmonids as umbrella species, i.e., species whose conservation confer a protective umbrella to numerous other co-occurring species.

The AUs were the same as those in the terrestrial habitats assessment, and for each we calculated an index of relative conservation value. The index had three components: the density of hydro-geomorphic features, local salmonid habitats, and the accumulative downstream habitats. That is, the relative value of a small watershed is based on: (1) the density of wetlands and undeveloped floodplains inside it, (2) the quantity and quality of salmonid habitats inside it, and (3) the quantity and quality of salmonid habitats downstream of it. Quantity and quality of habitats were assessed for eight salmonid species.

We examined relative conservation value from two perspectives that reflect a quantity versus quality dichotomy. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's single most significant contribution, i.e., the quality a place contributes. The two perspectives result in different rankings of AUs.

1.3.3. Marine Shoreline Habitats Assessment

In contrast to the terrestrial and freshwater assessments, the marine shoreline assessment had a large variety of comprehensive, reasonably accurate occurrence data for animals and plants. Given the quality of data for the shorelines of Puget Sound, we believed an assessment based on the presence of species would provide a credible indicator of relative conservation value. Hence, the overarching assumption of that decision is that the relative value of shorelines for the conservation of fish and wildlife habitats is mostly a function of the presence of the species for which we collect occurrence data. In general, we collect occurrence data for certain species because 1) humans harvest those species, 2) we are concerned about the status of those species (e.g. threatened or endangered species), or 3) we are concerned about the management of those species (e.g., species sensitive to human disturbances). In other words, we collect data on those species and habitats we care most about. Therefore, an assessment based on these data should indicate those places we should care most about for the conservation of fish and wildlife habitats.

The assessment units for marine shoreline habitats were small shoreline reaches with an average length of 0.24 miles, and for each we calculated an index of relative conservation value. The index had 41 components which included eight shellfish[4] species or species groups of commercial/recreational interest, urchins, three forage fish species, eight salmonid species, numerous bird species, pinnipeds, kelp, eelgrass, surfgrass, and wetlands. Like the freshwater habitats assessment, we examined relative conservation value from two perspectives that reflect a quantity versus quality dichotomy.

[4] Shellfish includes both mollusks such as butter clam and crustaceans such as Dungeness crab.

Part 2:
Terrestrial Habitats Assessment

2.1. Conceptual Model

A conceptual model is a simplified representation of a complex system that emphasizes the interrelationships of the major elements rather than the details of each element. The conceptual model is the basis for the components and structure of the quantitative model.

2.1.1. Scientific Foundation

The Department of Ecology's water flow assessment (Stanley et al. 2011) is based on the major watershed-scale hydrological processes that naturally govern stream flows. Unlike the water flow assessment, the conceptual model for the terrestrial habitats assessment is less process based and more function based. It is not process based because in the lower-elevation landscapes of the Puget Sound Basin, where city and county governments have principal jurisdiction over land use, the most important natural process for creating and maintaining terrestrial habitats no longer operates at a landscape scale. Prior to European settlement, wildfire, followed by natural regeneration and succession, were the processes that created and maintained a variety of forest and grassland habitats. The moist western hemlock forests of the western Cascades had a fire return interval between 200 and 750 years (Agee 1993). Stand-replacing fires occurred after periods of prolonged drought and burned over many thousands of acres. Wildfire also maintained prairie and oak woodlands habitats by suppressing encroachment of coniferous trees (Kruckeberg 1991). Over the past century, however, wildfire has been controlled for the purposes of protecting property and valuable forest resources, and consequently, fire has been effectively eliminated from the Puget Sound lowlands and Cascades foothills. Smaller-scale (on the order of ¼ to 100 acres) natural disturbances caused by wind or landslides still occur, but the dominant large-scale disturbances are now those related to human land uses (Figure 2.1).

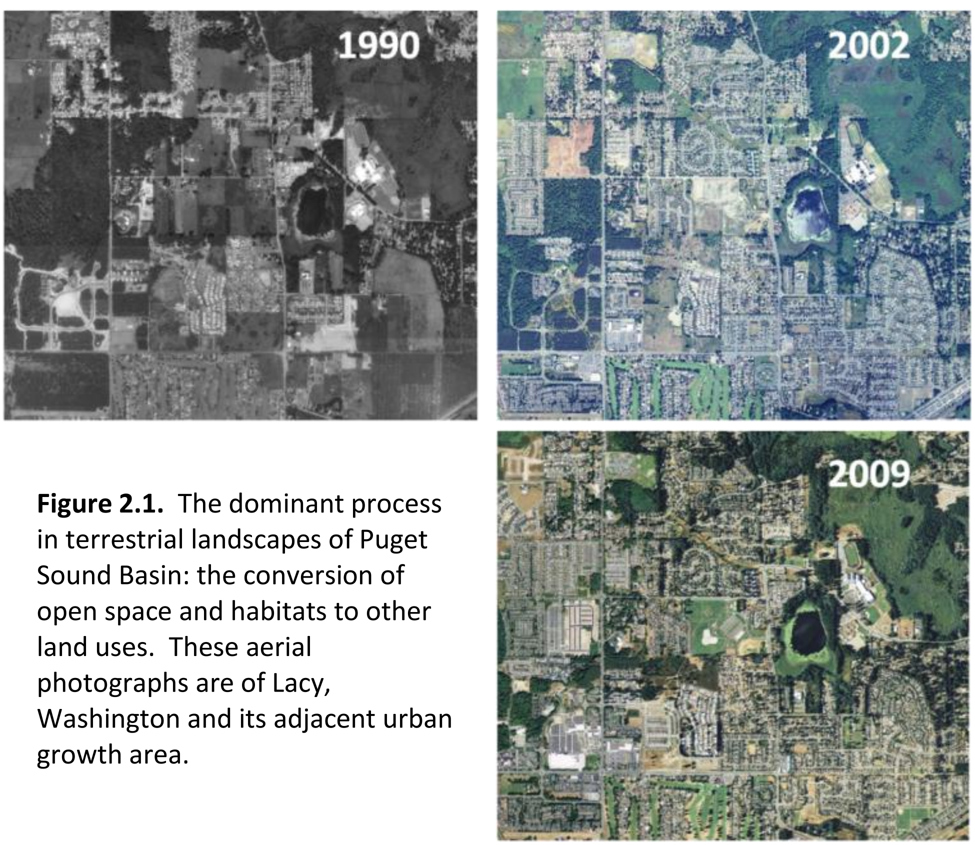


Figure 2.1. The dominant process in terrestrial landscapes of Puget Sound Basin: the conversion of open space and habitats to other land uses. These aerial photographs are of Lacy, Washington and its adjacent urban growth area.

Wildlife Communities

Successful conservation of native biodiversity may hinge on our ability to conserve natural biological communities (Karr 1990, Olden 2003). A biological community is an assemblage of species populations with a composition and structure determined by characteristics of the environment and by the relationships of each species to every other species. Each community exhibits emergent properties that transcend those of the individual species comprising them (Pianka 1988, Wilson 1997). Communities represent higher-level units of biodiversity that should be conserved for their unique qualities and ecosystem functions.

From an ecosystem perspective, a species' habitat is more than the abiotic and vegetative components of its environment. A species' habitat also consists of its biological community − i.e., all the other species with which it interacts. Hence, altering the species composition or structure of a community may dramatically alter some inter-specific relationships, degrade species-specific habitat quality, and lead to adverse consequences for particular species (Mills et al. 1993, Worm and Duffy 2003). Successful conservation of native wildlife in the Puget Sound Basin will depend on our ability to conserve natural wildlife communities. This is one reason our conceptual model emphasizes wildlife communities rather than individual species.

All land uses support a wildlife community, however, the species composition of each community can be dramatically different among land uses (McKinney 2002, Hansen et al. 2005). Industrial, commercial, and urban residential land uses tend to favor exotic species, such as house sparrow[5], European starling, rock pigeon, eastern gray squirrel, and Norwegian rat, or common synanthropic[6] native species such the American crow, Canada goose, house finch, American robin, and raccoon. Suburban residential land use favors many of these same species but also provides habitats for species associated with edges and early-successional forest, such as the song sparrow, white-crowned sparrow, American goldfinch, cedar waxwing, northern flicker, black-tailed deer, and red fox. In contrast, species associated with interior or conifer forest, such as brown creeper, Swainson's thrush, winter wren, Pacific-slope fly catcher, pileated woodpecker, and Douglas' squirrel are commonly found in commercial forests but are uncommon or rare in urban and suburban residential areas. Likewise, species associated with prairies and oak woodlands – such as white-breasted nuthatch, western bluebird, and horned lark − are not found in urban or suburban areas.

A major premise of our assessment is that different land uses support different wildlife communities. Industrial, commercial, and urban land uses support communities dominated by exotic species. Suburban residential areas support communities dominated by edge and early-successional forest species. Commercial forestry can support communities comprised of early, mid, or late successional forest species, depending on the management regime. We assumed that the capacity of a particular land use to support a natural wildlife community is determined by how the environment (i.e., the abiotic and vegetative components) created by that land use resembles the historical (i.e., pre-1850) environment. Hence, another key to successful wildlife conservation in the Puget Sound Basin is to identify those places that most closely resemble the historical environment, and then maintain or restore habitat conditions at those places.

---

[5] Scientific names of animals listed in Appendix E.

[6] Synanthropic species − wild animals or plants that live near and benefit from an association with humans and the somewhat artificial habitats that humans create around them; a form of symbiotic commensalism.

Landscape Integrity

One measure of how closely a place resembles the historical environment is ecological integrity. *Ecological integrity* is the ability of an ecological system to support and maintain a biological community that has species composition, diversity, and functional organization comparable to those of natural habitats within a region (Parrish et al. 2003). An ecological system has integrity when its dominant ecological characteristics (e.g., composition, structure, processes, and functions) occur within their natural ranges of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human disruptions.

Ecological integrity is the degree to which ecological structures, processes, and functions are complete and unimpaired. Ecological integrity is a vague concept. Its meaning is much discussed in the scientific literature, but there is no generally accepted operational definition (Quigley et al. 2001). Ecological integrity encompasses ecosystem health and stability. The concept describes systems that are whole and intact (Andreasen et al. 2001), and is related to the concepts of "human footprint" (Leu et al. 2008) and "naturalness" (Theobald 2010). Ecological integrity is a multi-scale concept, and can be assessed at a stand (or site) scale, landscape scale, and scales in between. Because this is a regional assessment covering a huge spatial extent, we assessed only landscape-scale ecological integrity, which we henceforth refer to as landscape integrity.

Landscape integrity is assessed over large spatial extents ($\sim 10^5$ acres) using spatial data such as roads, land use, land cover, housing density, or human population density that serve as surrogates for adverse changes to native habitats or impacts to ecosystems (Brown and Vivas 2005, Leu et al. 2008, Theobald 2010). Functional relationships between these surrogates and landscape integrity are most often formulated through expert judgment (e.g., Quigley et al. 2001, Mattson and Angermeier 2007), and hence, usually do not explicitly incorporate any known empirical relationships between biological responses and these surrogates.

One surrogate, among others, that we used to assess landscape integrity was land use. Land use has been demonstrated to affect ecological integrity (e.g., Glennan and Porter 2005). Our conceptual model divides land use into the following five major types: forestry, agriculture, residential, commercial-industrial, and public natural resources. All land uses except commercial-industrial were assumed to have a potential to be "open space." We defined open space as an area containing natural or semi-natural habitats or that functions as habitats for native wildlife. Under this definition, all open space is assumed to serve some habitat functions, however, the types of habitat functions and quality of those functions are dependent on the land uses within the open space.

Among the four major types of land use that may contribute to open space, residential land uses, in general, have the greatest negative impact on landscape integrity. Nevertheless, residential land uses support diverse wildlife communities. The species composition, richness, and evenness of these communities are correlated with dwelling density (Hansen et al. 2005, Donnelly and Marzluff 2006). Many native species become less abundant as dwelling density increases and some native species cannot tolerate even moderate density development (1 to 2 dwellings per 5 acres; Azerrad et al. 2009). Very low housing density (< 1 dwelling per 20 acres) results in *de facto* open space, and if native habitats are retained, then open space in residential areas may contribute substantially to the conservation of natural wildlife communities (Wilhere et al. 2007).

After residential land uses, agricultural land uses have the next greatest negative impact on landscape integrity. Certain agricultural land uses, however, possess characteristics resembling native habitat types, and consequently, provide habitat for species associated with those habitat types. For example, pastures in Thurston County provide habitat for western pocket gophers that are usually associated with native prairies. Agricultural land uses that don't resemble native habitat types can also provide high-value habitats for certain species. For instance, elk graze in pastures, and residual grain in harvested fields provide winter-feeding and resting areas for snow geese and other migratory water fowl. In both instances, the agricultural land use enhances the productivity and abundance of these species.

Forestry is an exception to the habitat loss or degradation that occurs through most other types of human land-use. Private commercial forests of the Puget Trough Ecoregion are mostly 2nd, 3rd and even 4th growth forests managed for timber. Forestry can, to a limited degree, mimic some aspects of natural disturbance and succession at both stand and landscape scales. And although typically depauperate in the key structural components (such as large trees, large snags, and large logs) of late-successional forests that historically dominated the Puget Sound lowlands and Cascades foothills, private commercial forestry can support wildlife communities with many of the same vertebrate species that comprise natural wildlife communities. Different landscapes comprised of private commercial forests are relatively similar in character but there are stand-level differences due to differences in management practices.

Among all land uses, public lands dedicated to conserving natural ecosystems, such as wilderness areas and national parks, have the highest landscape integrity, most closely resemble the historical environment, and hence, are most likely to support natural wildlife communities. Public lands managed for multiple-uses, such as state and national forests, generally possess less landscape integrity than national parks. Consequently, the species diversity (composition, richness, and evenness) of wildlife communities inhabiting state and national forests may be quite different than the wildlife communities inhabiting areas with higher landscape integrity. Relative to managed forests on public lands, wildlife communities in private commercial forests are likely to be even more divergent from wildlife communities found in wilderness areas or national parks.

Another surrogate we used to assess landscape integrity was habitat fragmentation. Numerous empirical studies have established that habitat fragmentation affects wildlife communities (Fahrig 2003). There at least 90 different metrics for describing various aspects of habitat fragmentation (McGarigal and Marks 1995). However, most indices are functions of patch size, patch shape, patch isolation or some combination of the three. When assessing habitat fragmentation, size matters. In fact, the area of contiguous habitat is likely the single most important patch property determining the long-term viability of wildlife populations (Diamond 1975, Fahrig 2003). Shape is a property the affects internal connectivity (Diamond 1975) and the severity of edge effects (Saunders et al. 1991). Compact shapes, such as circles or squares, which minimize the perimeter to area ratio, are better than more elongated or irregular shapes. The importance of shape is a function of patch size – for large patch sizes, patch shape may have little or no effect on wildlife populations (Saunders et al. 1991). Patch isolation is a measure of the ability of individual organisms to move among patches. Isolated patches are less likely to exchange individuals with other patches, and hence, species within isolated patches are more likely to become locally extinct. Isolation is a function of distance between patches and the size of patches. The proximity index of Gustafson and Parker (1994) has been shown to be a robust and reliable metric for patch isolation (Bender et al. 2003).

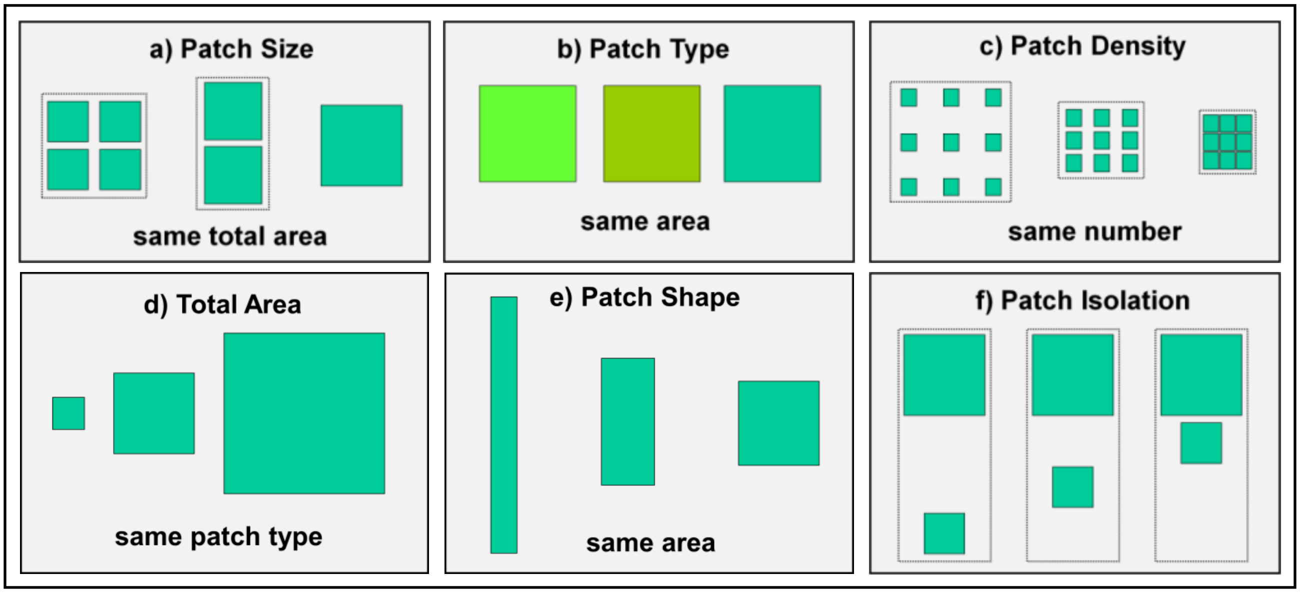


Figure 2.2. Landscape-level characteristics commonly used to assess landscape integrity. Landscape integrity improves from left to right in each gray box. We used patch size, patch shape, patch isolation, and patch type to assess landscape integrity. Patch type refers to the land use within a patch.

2.1.1. Modeling Relative Conservation Value

We developed an index that quantifies the relative value of places for the conservation of terrestrial wildlife communities. The principal challenge we faced were the limitations imposed by the currently available spatial data. Data on the locations of wildlife species collected by WDFW and other agencies generally focus on imperiled species or harvested species, and consequently, we have reasonably accurate data across the entire Puget Sound Basin for only a small number of animal species. To compensate for the lack of data we utilized a “coarse filter-fine filter” approach. This approach (*sensu* Noss 1987) divides all species into two groups: coarse filter and fine filter. Coarse-filter species are those that can be conserved by focusing on the conservation of habitat types. In theory, effective conservation of a habitat type should also conserve the wildlife community inhabiting that habitat type. Fine-filter species are those we believe cannot be conserved by conserving habitat types alone. Fine-filters species are usually rare or imperiled or species with special habitat requirements.

Our coarse filter-fine filter approach was somewhat unconventional. We were led to this unconventional approach by: (1) the assessment’s main application − local land use zoning; (2) the coarse spatial grain (~5 mi^2) and huge spatial extent of the assessment; (3) the low classification accuracy (35 % error) of the available spatial data that could be used to map habitat types (Sanborn 2007); (4) the lack of any reasonably accurate spatial data for the age or structural condition of habitat types; and (5) the relatively homogenous management of native habitats on low-elevation (<2800 ft) private lands in the Puget Sound Basin. Given these facts, we believed that a coarse-filter assessment based on a detailed mapping of habitat types would be unnecessary and inaccurate. Hence, our coarse-filter was simply landscape integrity. We assumed that identifying and conserving areas with the greatest landscape integrity should effectively conserve the extant natural terrestrial wildlife communities of the Puget Sound Basin.

It's worth repeating that our decision to use landscape integrity as a coarse filter was based on the scale of the assessment and the quality of available spatial data. Given the coarse scale of the intended application (i.e., local land use zoning), stand-scale differences among forests are irrelevant. Furthermore, with available spatial data, we could not accurately discern age class or structural differences among forests at the stand scale. Hence, at the stand scale, all forests of similar age on private commercial timberlands were assumed to have equivalent conservation value for terrestrial wildlife. For the purposes of local and use planning, the important distinctions among forests occur at the landscape scale, and one key distinguishing landscape-scale characteristic is landscape integrity. We also lacked the spatial data needed to make distinctions among different types of agriculture, such as pasture, orchard, and row crops. Consequently all agricultural lands were treated as equivalent.

An axiom of economics is that scarcity determines value. In the Puget Sound Basin, landscape integrity is scarcer at lower elevations than at higher elevations, and hence, landscape integrity is more valuable at lower elevations. Landscape integrity varies across vegetation zones because of differences in the amount of habitat loss, habitat fragmentation, and degrees of habitat protection. For instance, 89% of high elevation vegetation zones in Puget Sound Basin (2.9 million acres) have some level of protection on public lands. However, only 11% of low elevation zones in Puget Sound Basin (Oak Woodland-Prairie Mosaic, Puget Sound Douglas-fir, and Sitka Spruce) are protected on public lands. Low elevation zones contain imperiled habitats such as oak woodlands and prairies, uncommon habitats such as stands of mixed Douglas-fir and madrone, and biologically rich and productive habitats such as large wetland complexes and river floodplain forests.[7] Habitats in low elevation vegetation zones are more at-risk than habitats in higher elevation zones because of current land uses, potential future land uses, or private-public ownership patterns. For this reason vegetation zone (Figure 2.3) was a factor used to influence the relative conservation value of places. In other words, places with high landscape integrity in low elevation zones were considered more valuable than places with equivalent landscape integrity in higher elevation zones.

Landscape integrity alone cannot identify all high value places for the conservation of terrestrial wildlife species. Hence, we included fine-filter elements too. The fine-filter elements in the assessment were priority species and habitats as designated by WDFW's Priority Species and Habitats program (PHS; WDFW 2008). Priority species require protective measures for their survival due to their population status, sensitivity to habitat alteration, and/or recreational, commercial, or tribal importance. Priority species include State Endangered, Threatened, Sensitive, and Candidate species; and animal aggregations considered vulnerable (e.g., heron colonies, bat colonies). Priority habitat types are those with a unique or significant value to a diverse assemblage of species.[8]

Much of the PHS data are site-scale (e.g., nest and den sites), which does not match the scale of the assessment. Most site-scale occurrences are currently addressed by site-level management, such as critical area ordinances. This assessment is intended for landscape-scale land use planning. Hence, we used only PHS data that were landscape-scale occurrences, defined as occurrences greater than 10 to

[7] Wetland habitat types are covered in the freshwater and marine shoreline habitats assessments.

[8] A priority habitat may consist of a unique vegetation type (e.g., prairie) or dominant plant species (e.g., oak woodland), a described successional stage (e.g., old-growth forest), or a specific habitat feature (e.g., cliffs). With the exception of oak grassland prairie, we did not have comprehensive accurate mapping for any other priority habitats.

100 acres in size, depending on the species (Table A5). We also wanted data that were comprehensive, i.e., data that represented nearly all extant locations (> 85%) for that species in the Puget Sound Basin.

Our coarse filter-fine filter approach resulted in a model of conservation value consisting of two main components: 1) landscape integrity, and 2) the presence of PHS species or habitats (Figure 2.4). *Relative* landscape integrity was a function of open-space fragmentation and land use impacts. Separate indices of fragmentation and land use impacts were combined to create an index of landscape integrity.

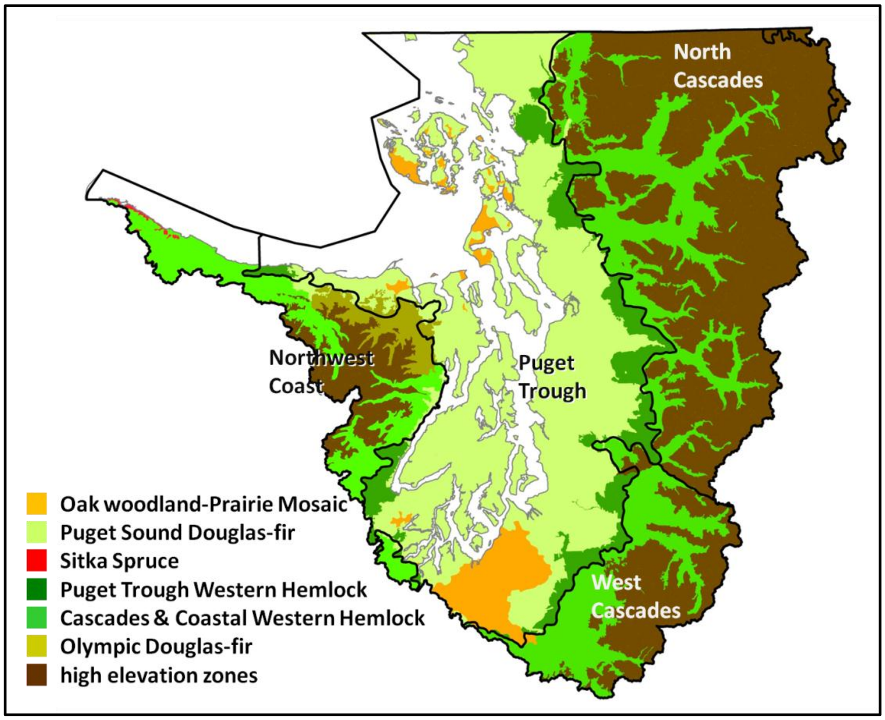


Figure 2.3. Vegetation zones of the Puget Trough Basin (modified from Cassidy et al. 1997). Six high elevation zones were lumped into one zone. Thick black lines are boundaries of the four ecoregions that intersect the Puget Sound Basin.

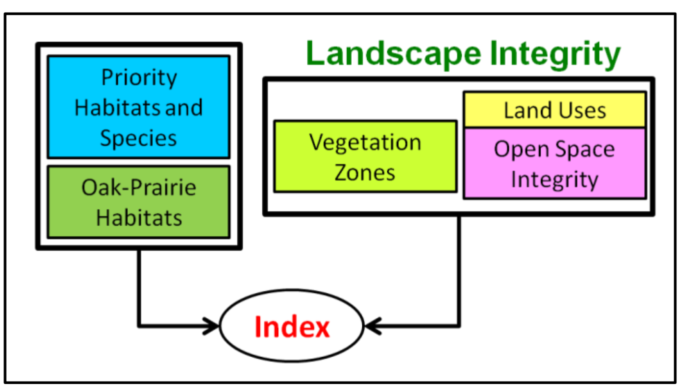


Figure 2.4. Major components of the terrestrial index of relative conservation value. Left branch consists of fine filter species and habitats. Right branch is effectively a coarse filter that identifies places with high landscape integrity.

2.2. Methods

This section describes the individual components of the terrestrial assessment and how they are assembled to yield an index of relative conservation value. The relative conservation value of watershed-based assessment units (AUs) was calculated in three stages. First, open-space blocks were identified. Second, the landscape integrity of open-space blocks was assessed. In the third stage, the landscape integrity of open-space blocks was combined with PHS habitats located in each AUs to yield an index of relative conservation value. More detailed explanation is presented in Appendix A.

2.2.1. Open Space Blocks

An open-space block is a contiguous area containing land uses – such as commercial forest, agriculture, parks, and designated open-space – that maintain natural or semi-natural habitats or serve as habitats for native wildlife. Three spatial data layers were used to identify open-space parcels: the Washington State Parcel Database developed by the Rural Technology Initiative (RTI 2011), land cover data developed by WDFW (Pierce 2011) using aerial photography from the National Agriculture Imagery Program, and the Major Public Lands layer created by the Washington Department of Natural Resources.

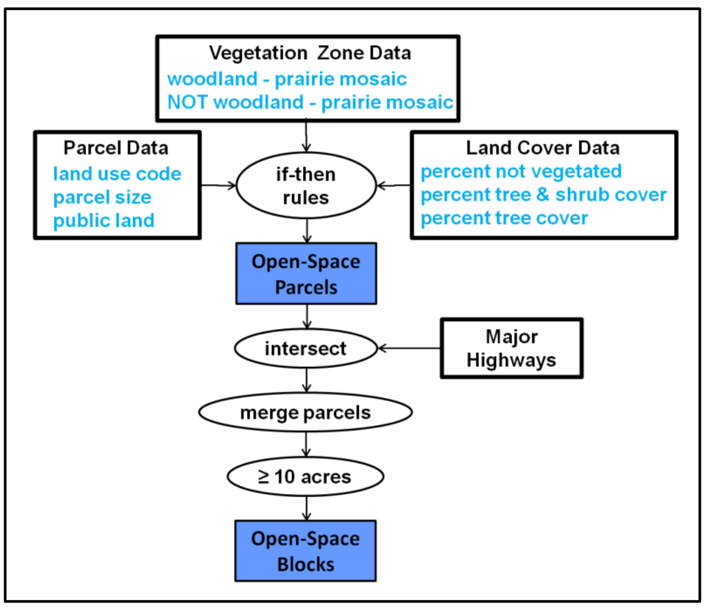


Figure 2.5. Process used to construct open-space blocks. Land cover data were from Pierce (2011), parcel data were from the Rural Technology Initiative (RTI 2010) at the University of Washington, and vegetation zone data were modified from Cassidy (1997). If-then rules presented in Table A2.

The Washington State Parcel Database contains the land use for all private land parcels in Puget Sound Basin (Table A1). We grouped the land uses into seven general categories: commercial-industrial, residential, agriculture, forestry, mining, mixed-use open space, designated open space. All categories except commercial-industrial can contribute to open space. For each general category we constructed rules that classified parcels as open space or not open space (Table A2). Rules consisted of three variables − parcel size, land cover, and vegetation zone – and were developed through an iterative process that relied on expert judgment. In the Washington State Parcel Database, data for state and federally managed public lands are missing or inconsistent. Consequently, for state and federally managed public lands we used the Major Public Lands spatial data layer.

The non-open space parcels were removed from the parcel database and major highways (state routes, federal and interstate highways) were intersected with the remaining parcels. This intersection split some parcels into smaller polygons. Boundaries between adjacent parcels were dissolved to form larger polygons and only polygons greater than 10 acres were retained as the final set of open-space blocks Figure 2.5).

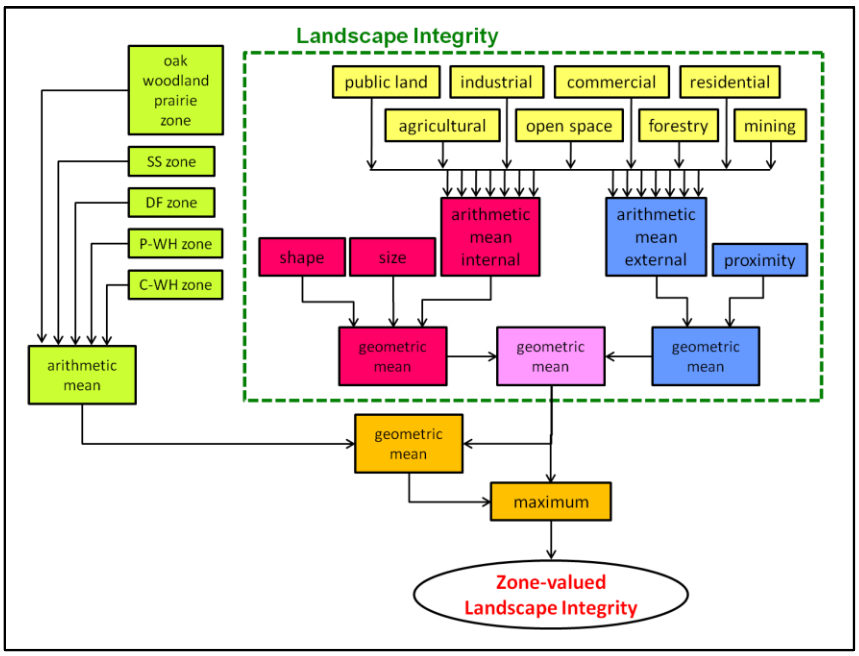


Figure 2.6. Model structure for zone-valued landscape integrity index applied to open-space blocks. Proximity and shape indices were calculated with FRAGSTATS (McGarigal and Marks 1995). Vegetation zones refer to the proportion of an open-space block intersecting each zone. Vegetation zones influence the value of landscape integrity to yield a zone-valued integrity index. The maximum function does the following: if the integrity index is high, then vegetation zone will not reduce it, but if integrity index is low, then vegetation zone can enhance it. In other words, integrity is a more important factor than vegetation zone.

2.2.2. Landscape Integrity of Open Space Blocks

The index of relative landscape integrity was based on expert judgment. Relative landscape integrity of each open-space block was a function of land use impacts and open-space fragmentation. Three spatial data layers were used to assess landscape integrity: the Washington State Parcel Database (RTI 2011), land cover data developed by WDFW (Pierce 2011), and the open-space blocks described in the preceding section. The parcel data give the main land use of every parcel. Different land uses have different degrees of adverse impact on landscape integrity. Each parcel's land use was assigned a potential adverse impact value from 1 to 1000 (low to high; Table A1). Impact of each parcel within an open-space block was weighted by parcel area. A weighted arithmetic mean of the impact values for land uses within the block served as an indicator of the internal impacts upon landscape integrity. A weighted mean land use impact was also calculated for land uses surrounding each block.

Metrics of open-space fragmentation were calculated for each open-space block using the program FRAGSTATS (McGarigal and Marks 1995). The indices used in the landscape integrity index were the shape index called "Circle" and the proximity index (Gustafson and Parker 1994). The equation for landscape integrity is shown schematically in Figure 2.6 and weights are summarized in Table A4.

Vegetation zone was also a factor used to influence the value of open-space blocks. Our vegetation zones were based on the GAP vegetation zones (Cassidy 1997). For each open-space block, an average vegetation zone value was calculated based on the area of each vegetation zone intersecting the open-space block and the relative value assigned to each vegetation zone. The relative value of each vegetation zone was a subjective judgment based on the percent of historical area lost and rarity of the zone. The equation for combining landscape integrity and vegetation zones to yield a *zone-valued landscape integrity* is shown schematically in Figure 2.6.

2.2.3. Species and Habitat Based Indices

We applied two filters to the PHS data: (1) the occurrence data for a species had to be landscape-scale, defined as occurrences greater than 10 to 100 acres in size, depending on the species; and (2) the data for a species had to represent nearly all habitat (> 85%) for that species in the Puget Sound Basin. These two filters limited the PHS data to 12 species represented by 441 polygons ranging in size from 10 to 1 million acres (Figure A5).

We also included two PHS habitat types, westside prairie and Oregon white oak woodlands, which were lumped into one habitat type − an oak-grassland type. The oak-grassland habitat type is perhaps the most imperiled terrestrial habitat type in the Puget Sound Basin. The Washington Natural Heritage Program has mapped prairie and oak woodland types with about the same degree of accuracy and precision as the PHS data we included in our assessment. Hence, we included the Heritage Program's oak-grassland habitat data and treated it in the same way as PHS data.

We developed a simple index for PHS habitats. For each AU we calculated for all 12 species the percent of the AU covered by the PHS polygons of the species and the percent of the species' entire habitat in the Basin contained within the AU (Figure A6). These 24 numbers were then adjusted such that a percentage greater than a threshold, T, was set to 100 and percentages less than T were translated to a 0 to 100 scale. The rationale for this threshold is that an AU that is greater than 25% PHS habitats, for instance, or contains more than 25% of a species' habitat in the Basin is invaluable. The index was the maximum of these 24 adjusted percentages. The same process was applied the oak-grassland habitat type.

2.2.4. Relative Conservation of Assessment Units

We created one index of relative conservation value that was a function of three components: mean zone-valued landscape integrity index, the PHS index, and the oak-grassland habitat type index. The function was simply the maximum of the three components.

2.2.5. Sensitivity and Uncertainty Analyses

We conducted both sensitivity and uncertainty analyses on the index of relative conservation value. See Appendix A for detailed description.

2.3. Results

2.3.1. Open Space Blocks

We identified 7,640 open-space blocks which ranged in size from 10 acres to 1.3 million acres (Figure 2.7). Twenty-two percent of the blocks were greater than 100 acres but only three percent were greater than 1,000 acres. Nearly half the open-space area, 3.6 million acres, was encompassed by only 4 blocks which were located in the Olympic and Cascades Mountains and comprised mostly of public land.

Open-space blocks covered about 80% of the Puget Sound Basin. These open spaces are a mix of public and private lands, wilderness, parks, managed forest, agriculture, undeveloped or lightly developed residential parcels, and other types of open space. Open-space is not uniformly distributed in the Basin. Roughly half of all open space is located in high elevation zones and about 20 percent of open space is located in the low elevation Puget Sound Douglas-fir and Woodland-Prairie Mosaic zones. The high elevation zones are 99% open space and have an overall landscape integrity score of 0.98 (Figure 2.8).[9] The extraordinarily high landscape integrity score results from the national parks, wilderness, and roadless areas that comprise most of the high elevation zones. The Cascades and Coastal Western Hemlock Zones are 94% open space and have an overall landscape integrity score of 0.88. In contrast, the Puget Sound Douglas-fir zone is 49% open space and has an overall landscape integrity score of only 0.18. The lower elevation zones have much lower landscape integrity because the open-space blocks in these zones have smaller average area, are generally comprised of more negatively impacting agricultural and residential land uses, and are also surrounded by more negatively impacting land uses.

About 90% of the open space blocks had zone-valued landscape integrity scores less than 0.2 (Figure 2.9). However, these blocks comprise only about 3 percent of the total open space area that we identified. These lowest value blocks (zoned-valued landscape integrity < 0.1) tended to be less than 20 acres and surrounded by high impact land uses. Over 75% of the blocks' collective land area had zoned-valued landscape integrity scores over 0.9. The highest value blocks were either very large (>60,000 acres) or of moderate size and in a low-elevation vegetation zone, i.e., Puget Sound Douglas-fir or Woodland-Prairie Mosaic.

Spatial patterns of landscape integrity followed an expected pattern (Figure 2.10). Blocks with highest integrity were located in the Olympic and Cascades Mountains, which are dominated by large contiguous areas of managed and unmanaged forest. As one moves from higher elevations (>1500 ft) through the foothills toward the Puget lowlands, open-space blocks become smaller, are less forested, and are increasingly surrounded by more negatively impacting land uses such as agriculture and residential development. Upon reaching the lowlands, the vast majority of open-space blocks are between 10 and 50 acres and are farther apart. This separation results in more isolation between blocks. Many of the largest open-space blocks in the lowlands are dominated by agricultural land uses. Consequently, the lowest landscape integrity scores occur in the Puget lowlands.

[9] By convention landscape integrity and zone-valued landscape integrity have a maximum value of 1 and a minimum value of zero.

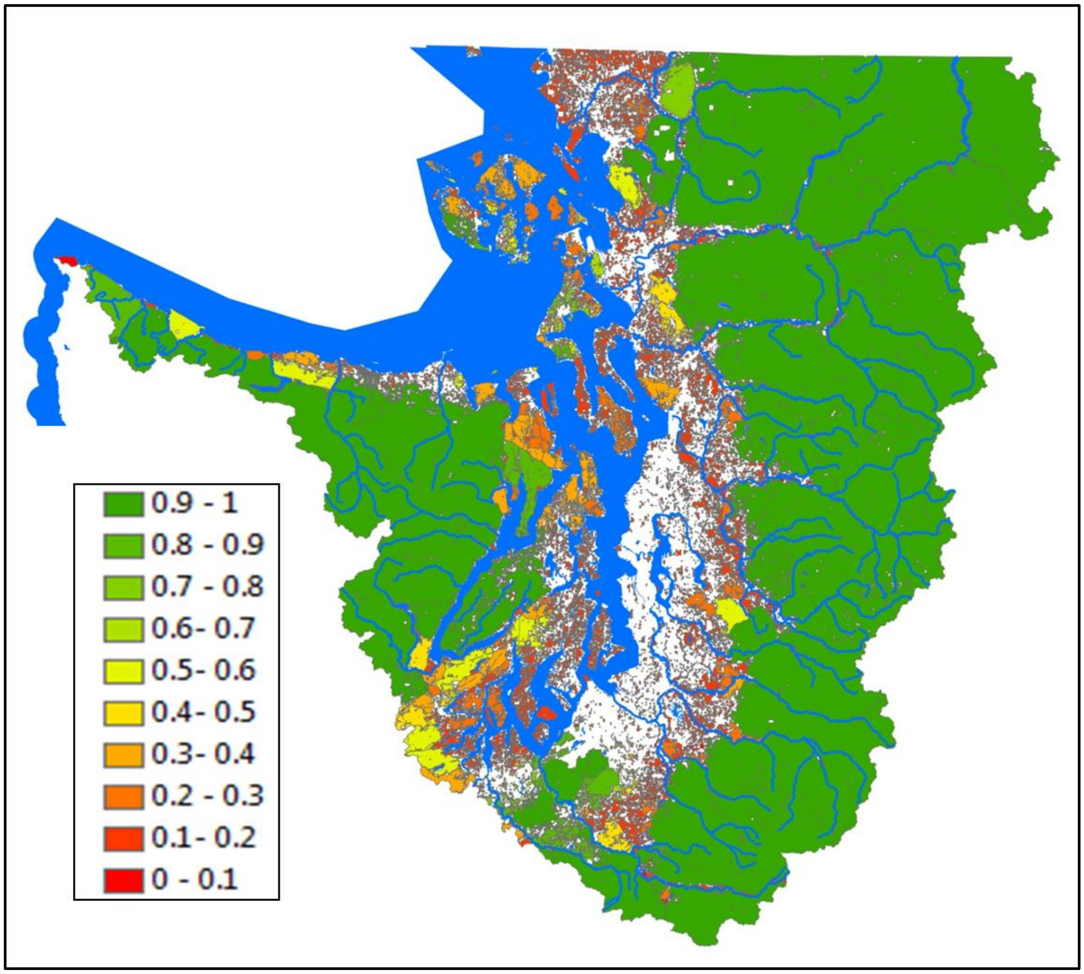


Figure 2.7. Zone-valued landscape integrity of open-space blocks for all of Puget Sound Basin. Highest value blocks are dark green and lowest value blocks are dark red. White space is not open space. Blue is water.

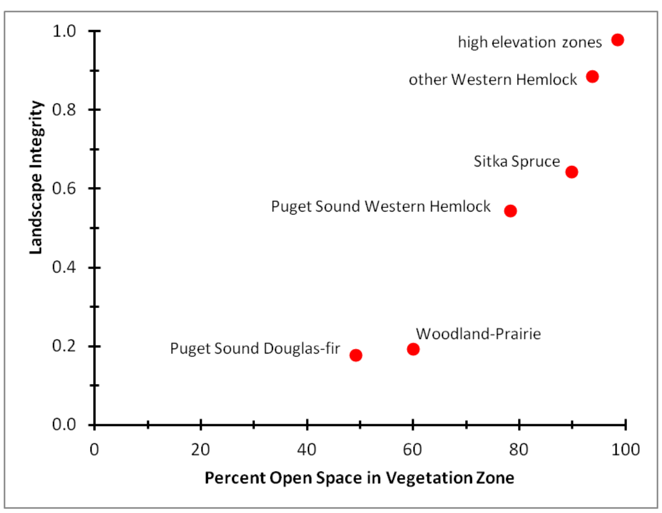


Figure 2.8. Percent area that is open-space and mean landscape integrity for seven of the eight vegetation zones used in the assessment. Cascades and Coastal Western Hemlock zones were combined to form “other” western hemlock.

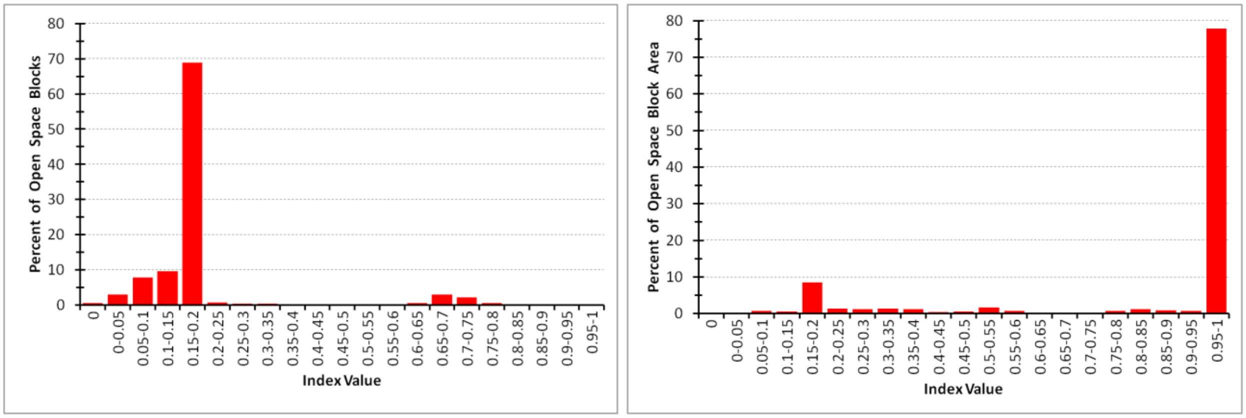


Figure 2.9. Distribution of zoned-valued landscape integrity by number of open-space blocks (left) and by land area of open-space blocks (right). There are 7,640 open space blocks in the Puget Sound Basin covering a total of 7,073,000 acres.

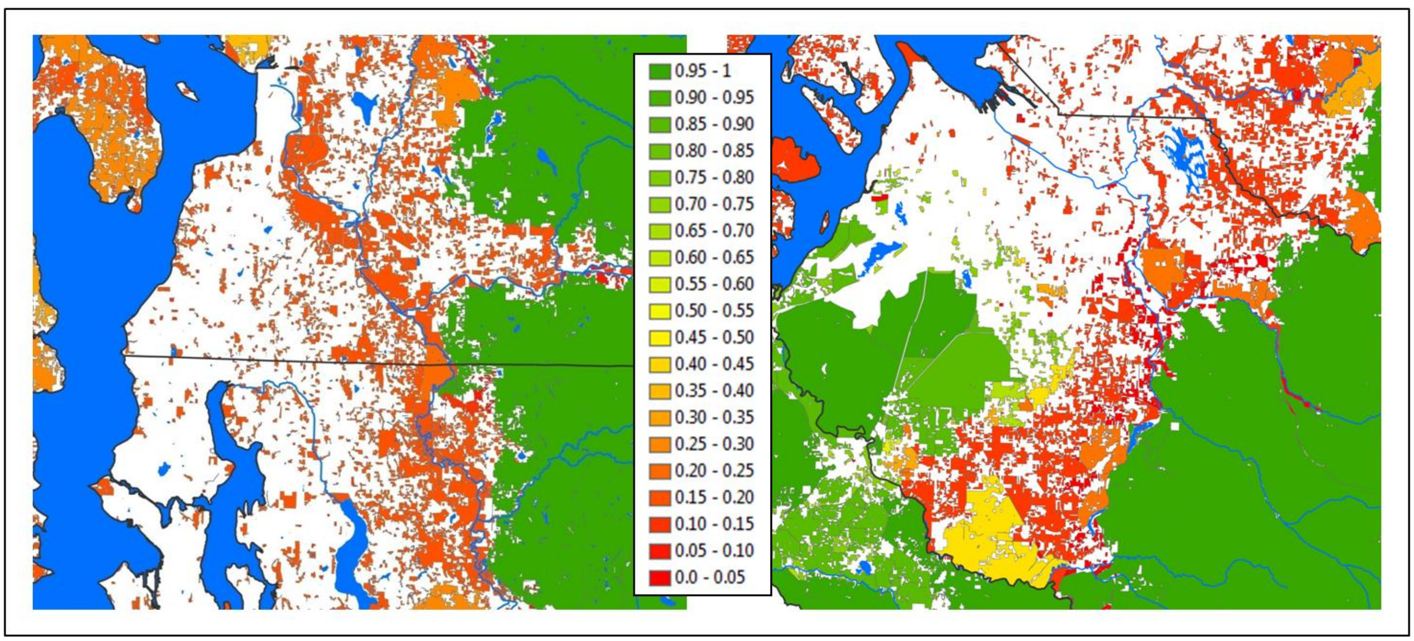


Figure 2.10. Zoned-valued landscape integrity of open-space blocks in two portions of the Puget Sound Basin: the Snohomish river valley (left) and western Pierce County (right). Highest zoned-valued landscape integrity is dark green and lowest integrity is red. White space is not open space. Black lines are county boundaries. Blue is water.

2.3.2. Relative Conservation Value

Of the 2940 AUs, about 37% had relative values greater than 0.9, 21% had relative values less than 0.1, and about 27% of the remaining AUs were evenly distributed between 0.2 and 0.9, (Figure 2.11). The highest value AUs either overlapped large open-space blocks (>60,000 acres) or contained a large proportion of a PHS habitat or the oak-grassland habitat type. The lowest value AUs overlapped a small number of small open-space blocks and contained no PHS habitat or oak-grassland habitat type. In terms of land area, about 63% of the Puget Sound Basin was in an AU that scored over 0.9. The vast majority of these AUs were in the Olympic or Cascades Mountains or their foothills and were mostly comprised of public lands.

The spatial pattern of AU conservation value (Figure 2.12) generally followed the spatial pattern of landscape integrity scores. Exceptions to this pattern occurred where the AU contained a large proportion of a PHS habitat or the oak-grassland habitat type, e.g., elk winter range near Sequim and bird overwintering areas near the mouths of the Snohomish, Stillaguamish, and Skagit Rivers (Figure 2.13). Recall that AU scores were the maximum of zone-valued landscape integrity, PHS, and oak-grassland indices. Landscape integrity was the maximum value for 77% of AUs, PHS was the maximum value for 22% of AUs, and oak-grassland for approximately 1.5% of AUs. Eighteen percent of the land area in the Puget Sound Basin had relative conservation value scores less than 0.2. Given that 89% of high elevation zones in Puget Sound Basin have some level of protection on public lands but only 11% of low elevation vegetation zones do, the spatial distributions of high and low landscape integrity are not surprising. What is surprising is the relatively small area of the Basin that had moderate conservation value according to our index. Only 15% of the Basin's land area was in AUs that had relative conservation value between 0.2 and 0.8.

The main application of the assessment is local land use plans affecting private lands. For those AUs with a substantial amount of private land (> 33% of the AU), 48% have relative conservation value less than 0.2 and about 20% have relative conservation value greater than 0.9 (Figure 2.14). In contrast, for those AUs with a substantial amount of public land (> 33% of the AU), relative conservation value is greater than 0.90 for 74 percent of AUs.

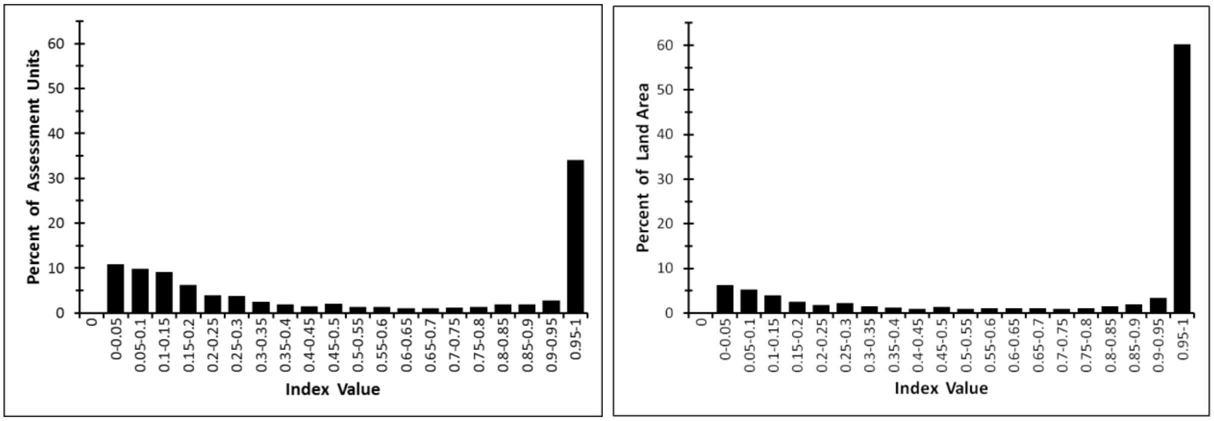


Figure 2.11. Distribution of conservation value index by number of assessment units (left) and by percent of land area in Puget Sound Basin (right). There are 2,940 AUs in the Puget Sound Basin.

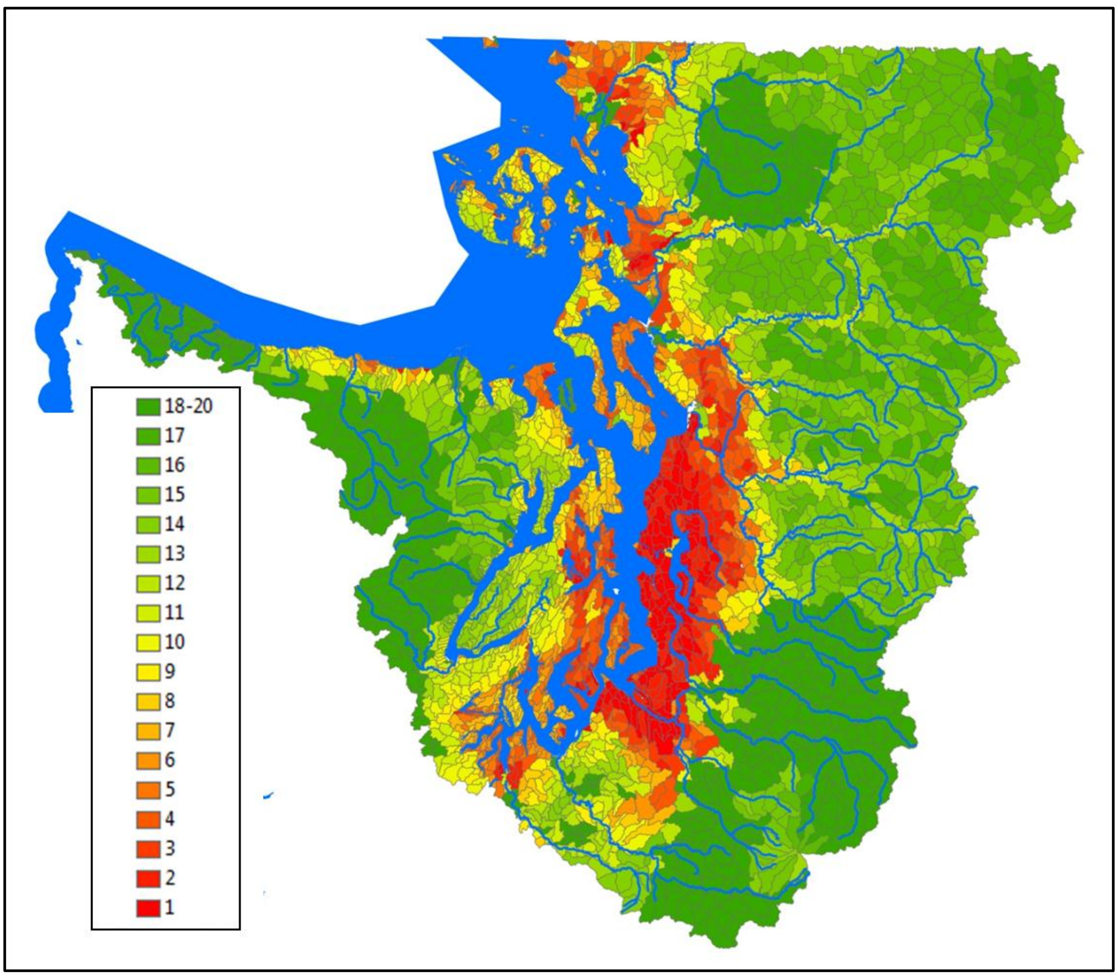


Figure 2.12. Relative conservation value for all assessment unit (AUs) in the Puget Sound Basin. Highest relative value AUs are dark green and lowest relative value AUs are dark red. Blue is water.

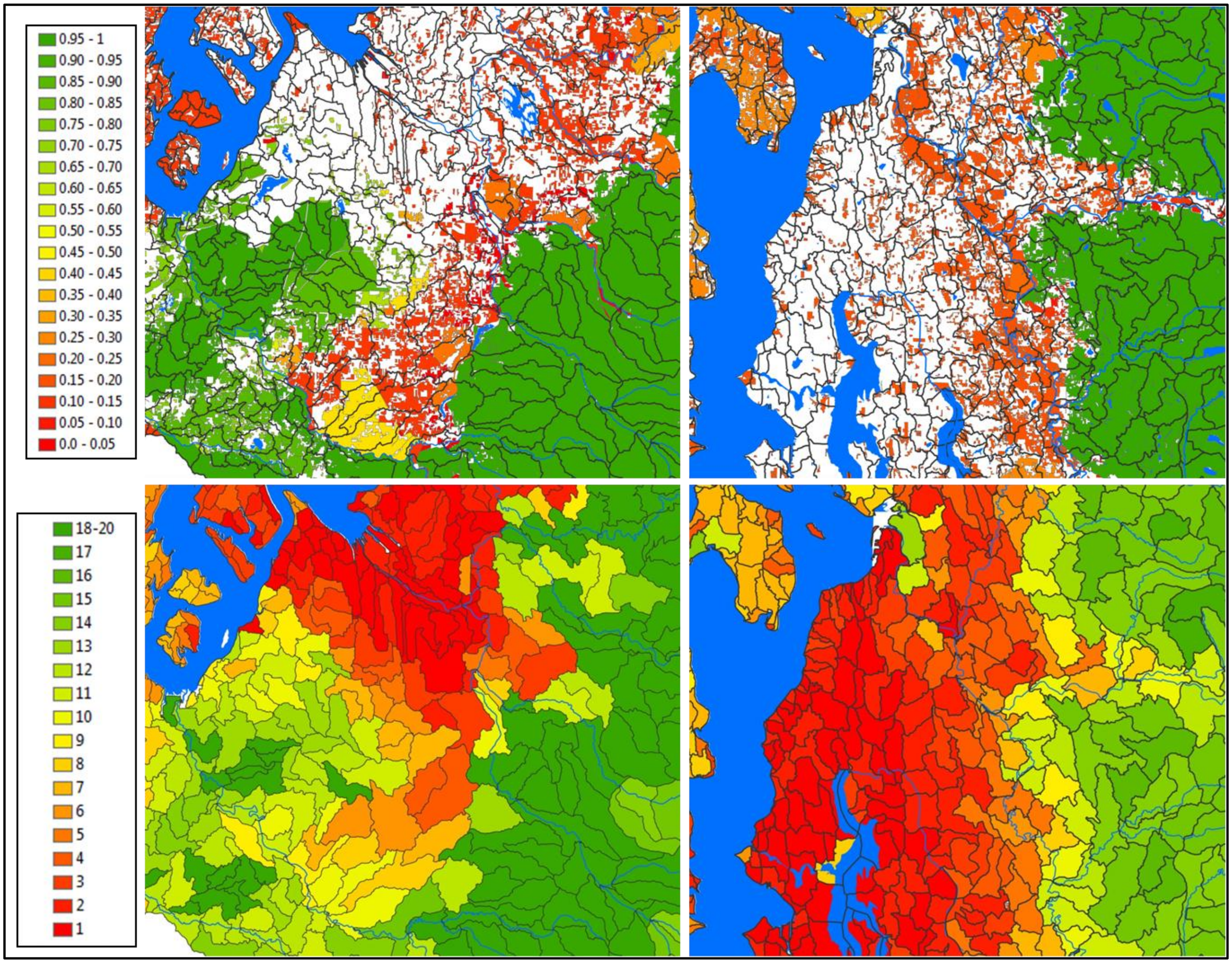


Figure 2.13. Assessment results in two portions of the Puget Sound Basin: the Snohomish River valley (left) and western Pierce County (right). Top panels show open-space blocks overlaid with AU boundaries. White space is not open space. Bottom panels show relative conservation value of AUs for the same area. Highest relative value AUs are dark green and lowest relative value AUs are dark red. Blue is water.

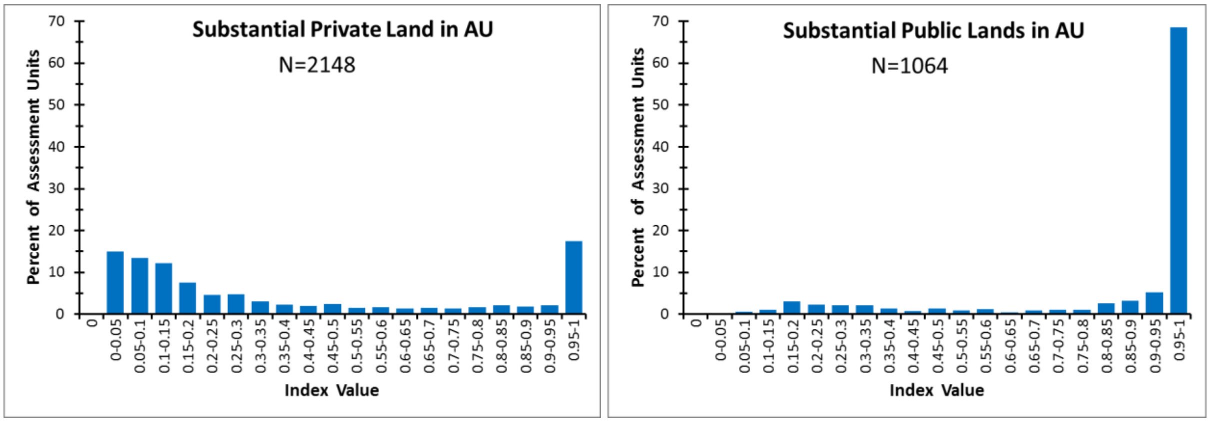


Figure 2.14. Distribution of conservation value index for AUs with a substantial amount of private land (left) and with a substantial amount of public land (right). "Substantial amount" was set to at least 33% of AU area.

2.3.3. Sensitivity and Uncertainty Analyses

Sensitivity Analysis

Mean average elasticity for all AUs collectively (Figure 2.15) showed that parameters with the greatest influence on relative conservation value are those that affect the most AUs. For instance, the model was most sensitive to the parameter determining the relative influence of the Puget Sound Douglas-fir zone because about 80% of the open-space blocks where located in that vegetation zone. Hence, a 5% increase in that parameter caused *relative* conservation value of many AUs to increase. Likewise, but for opposite reasons, the parameter determining the relative influence of the oak woodland-prairie mosaic zone had the largest negative influence because that zone contained the fewest open-space blocks. A 5% increase in that parameter caused the relative conservation value of a small number of AUs to increase, and hence, the *relative* conservation value of a large number of AUs decreased. In the model for landscape integrity, the size of open-space blocks and the land uses within the block (i.e., interior impacts) had the largest influence on relative conservation value.

Mean average elasticity for each AU individually (Figure 2.16) showed that as relative conservation value increases sensitivity to parameter changes decreases. In fact, most AUs with relative conservation value scores greater than 0.8 are nearly insensitive to changes in parameters. This indicates that scores above 0.8 are robust and unaffected by many of the assumptions of our model.

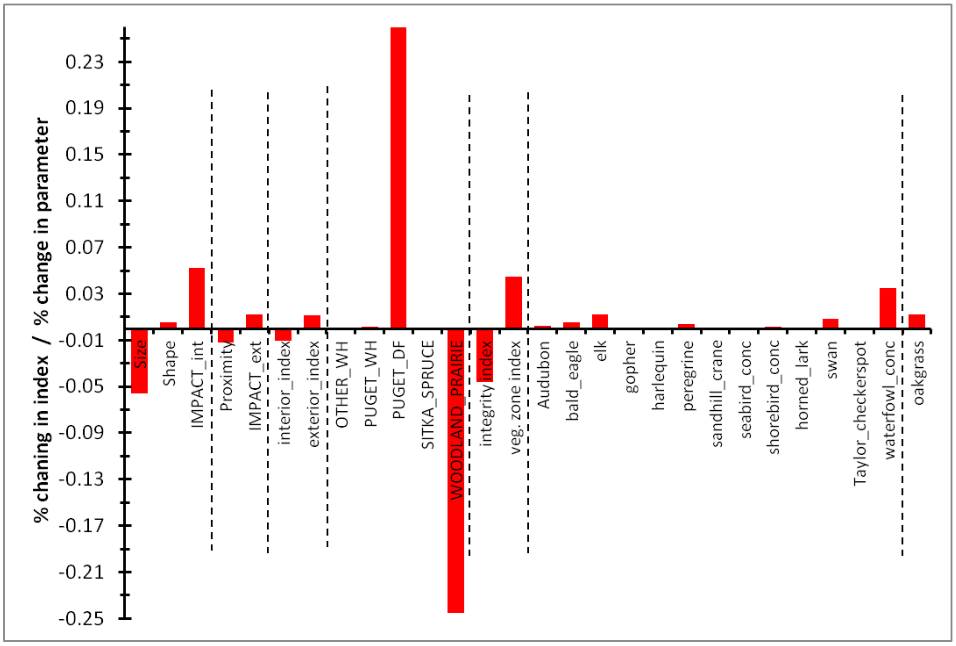


Figure 2.15. Mean average elasticity of relative conservation value to changes in model parameters for all AUs collectively. Parameters are the weights that determine the relative influence of variables used in calculating relative conservation value. Vertical dashed lines separate parameters that are in different components of the model. See Table A4 for description of weight parameters.

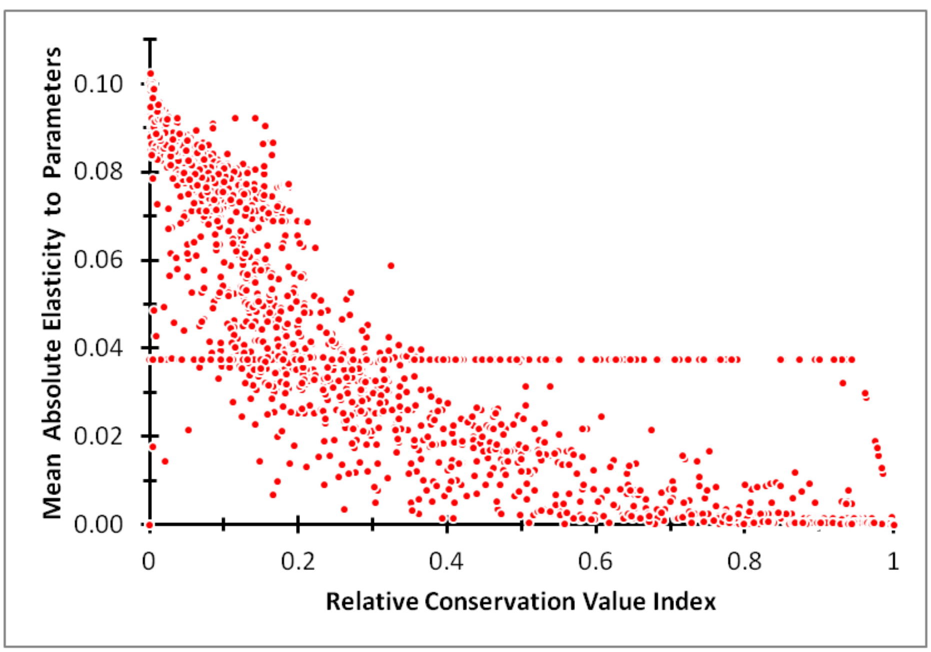


Figure 2.16. Mean average absolute elasticity of relative conservation value to changes in model parameters for each AU individually. Most AUs with index greater than 0.8 are effectively insensitive to changes in parameter values.

Uncertainty Analysis

The uncertainty analysis showed that relative conservation values near 1 or 0 have the narrowest 90% confidence intervals, i.e. have the smallest uncertainty (Figure 2.17). For scores near 1 the smallest uncertainties were for those AUs that contained a substantial amount of PHS habitat(s), e.g., AUs in the Olympic and Cascades Mountains containing elk habitat, and AUs near Puget Sound containing shorebird and waterfowl concentrations. For scores near 0, the smallest uncertainties occurred in urban areas that have very little open space and no PHS habitats. Uncertainty is generally greatest for AUs with conservation values between 0.3 and 0.8, but within that range some AUs had relatively narrow confidence intervals.

AUs with large uncertainty do not have less value. In fact, large uncertainty means that the actual relative conservation value could be larger or smaller than the value calculated. The 90% confidence intervals indicate that some AUs with relative conservation value as small as 0.5 could have relative conservation value over 0.90. This uncertainty must be taken into account when making land use plans.

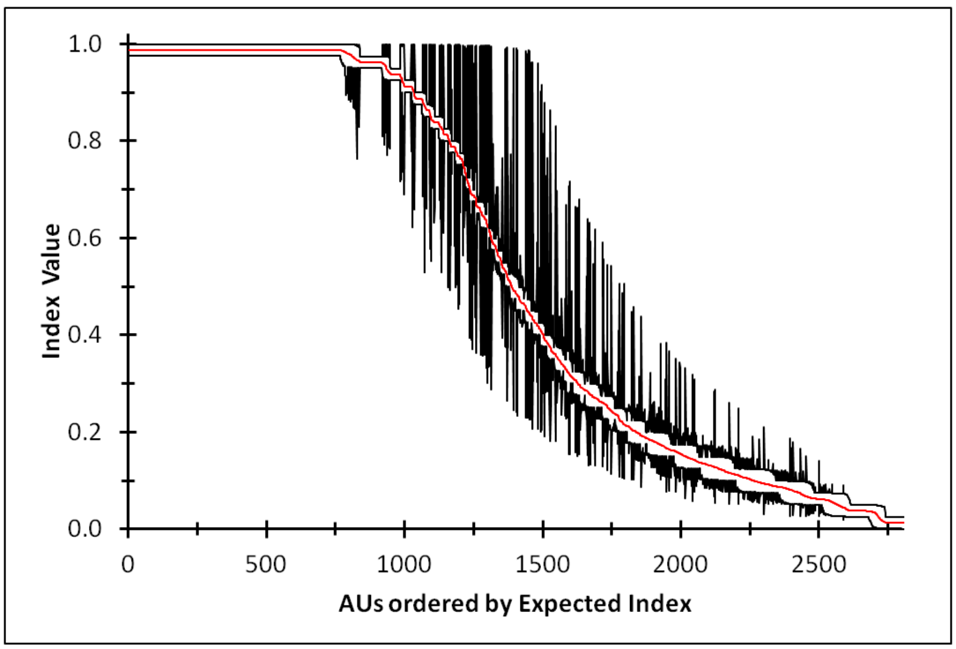


Figure 2.17. Uncertainty in AU relative conservation value. The 2,940 AUs are ordered from largest score to smallest score along the x-axis. The AUs' mean conservation values are plotted on the y-axis (red-line) along with the bounds of their 90% confidence internals (upper and lower black lines). Uncertainty is generally greatest for AUs with conservation values between 0.8 and 0.3.

2.4. Discussion

The main product of the terrestrial habitats assessment is a map that shows the relative conservation value of watershed-based assessment units throughout the Puget Sound Basin. The primary intended application of that map is land use planning done by local governments for comprehensive plan updates, sub-area plans, transfer of development rights programs, or other landscape-scale projects. City and county governments have regulatory authority over land uses within their jurisdictions, and the land use zones they designate through comprehensive or sub-area plans may be the most important actions affecting the health of terrestrial wildlife communities.

This assessment does not identify particular AUs that must be protected. The assessment is only a guide for landscape-scale habitat conservation. Local land use planning is governed by Washington's Growth Management Act (GMA). Under GMA, local land use plans must accommodate projected human population growth (RCW 36.70A.110, 36.70A.115). The terrestrial assessment should be used to direct new growth away from places with relatively high habitat value and toward places with relatively low habitat value (Figure 2.18). However, we recognize that "smart growth" will at times require compromises among multiple worthwhile but conflicting societal objectives.

Commercial forest and agricultural lands, collectively known as working lands, can meet multiple societal objectives. These working lands, particularly commercial forests, provide habitats for native species, valuable commodities, and ecosystem services. As the spatial distribution of land uses in the Puget Sound Basin changes over time so do the composition and structure of wildlife communities. Land use zoning that maintains or expands the current area of working lands may be the most effective action local governments can take for maintaining the health of wildlife communities in Puget Sound Basin.

Relative conservation value was based largely on landscape integrity. Our primary assumption was that places with higher landscape integrity are more likely to support wildlife communities more similar to natural wildlife communities than landscapes with lower integrity. The most important factor affecting landscape integrity is the size of open-space blocks. Therefore, for the sake of wildlife communities in Puget Sound Basin, maintaining the size of open-space blocks, especially those over 10,000 acres, should be a major consideration in landscape-scale projects such as comprehensive plan updates, sub-area plans, and transfer of development rights programs. Maintaining large open-space blocks on the order several million acres is of utmost importance for large-bodied, wide-ranging species such as elk, black bear, cougar, and gray wolf.

The spatial extent over which an assessment is conducted affects one's interpretation of relative conservation value. Wildlife do not recognize geopolitical boundaries and most are unimpeded by watershed boundaries. Furthermore, for some wide-ranging species, population-level habitat needs can encompass landscapes of several million acres. Hence, the extent of our terrestrial assessment covered the entire Puget Sound Basin with no spatial sub-divisions, and therefore, valid comparisons can be made amongst AUs in different WRIAs or different counties.

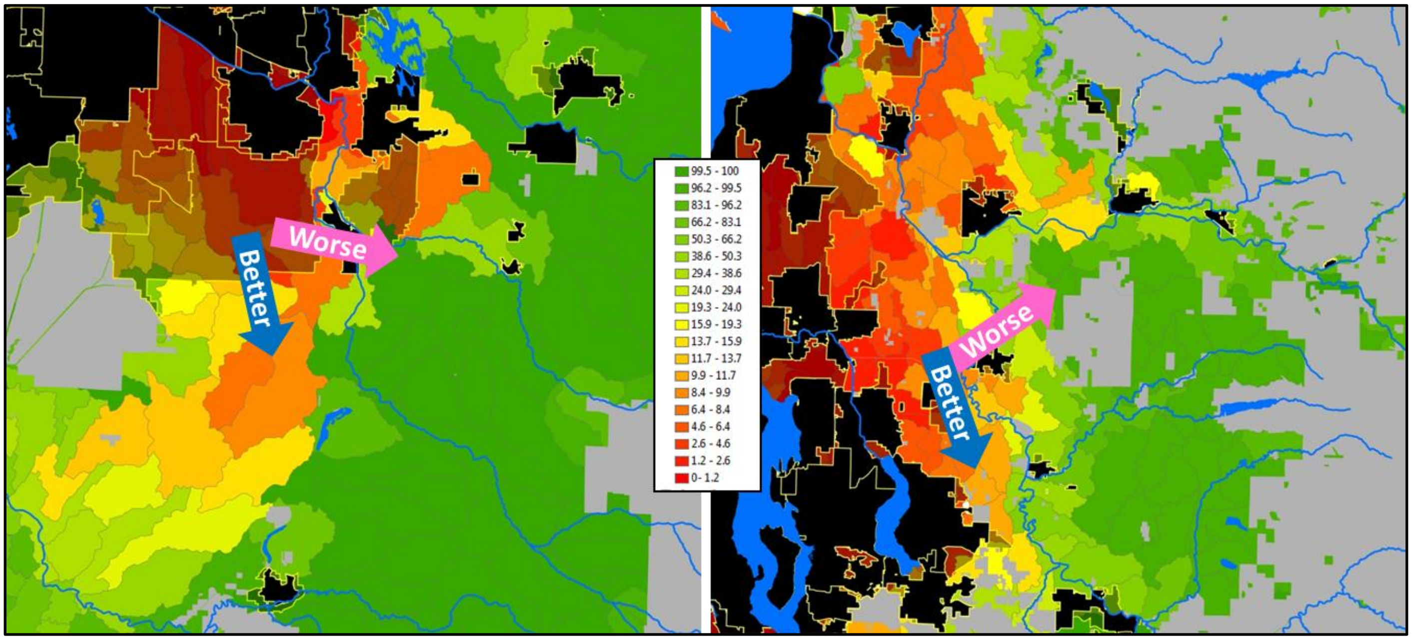


Figure 2.18. Two examples of using the terrestrial assessment results at the urbanizing fringe. Directing new human population growth to areas with low relative value (orange) is better for wildlife and directing new human population growth to areas with high relative value (green) is worse for wildlife. Examples are central Pierce County (left) and the Snohomish River valley (right). Opaque black areas are cities, translucent black areas are urban growth areas (UGAs), gray areas are public lands, and blue is water. Highest relative conservation value is dark green and lowest relative value is dark red. Scores are broken into 20 quantiles, i.e., groups containing 5% of AUs.

2.4.1. Validation

Validation entailed comparing the index scores against our collective knowledge of the Basin − did the index of relative conservation value show places we believed to be relatively more important as more important and places we believed to be relatively less important as less important. In nearly all places the index scores conform to our expectations. For instance, the foothills of the Olympic and Cascades Mountains have higher relative conservation value than Seattle and Tacoma. This difference reflects a gradient of relative conservation value from urban areas to wildlands that is repeated throughout the Puget Sound Basin. The exceptions to this pattern also conform to our expectations. For instance, our assessment shows that the mouths of major rivers, such as the Nisqually, Skagit, and Nooksack, which support large concentrations of waterfowl and shorebirds; and the oak-grassland habitat types in and around Fort Lewis have high conservation value.

Another form of validation is comparing our results to the results of other ecological assessments. During the past decade, one major effort has published maps depicting “high priority” sites for terrestrial habitat conservation in the Puget Sound Basin − the state-wide ecoregional assessments conducted by The Nature Conservancy and WDFW. The Puget Sound Basin overlaps four separate ecoregions: Pacific Northwest Coast, North Cascades, West Cascades, and the Georgia Basin-Puget Trough- Willamette Valley (GB-PT-WV). The ecoregion with the greatest overlap with the Puget Sound

Basin is the GB-PT-WV, which also encompasses most private lands in the Puget Sound Basin. That ecoregion was assessed by Floberg et al. (2004) and reassessed by Wilhere at al. (2008). Wilhere et al. (2008) used different methods than our terrestrial assessment. Wilhere et al. (2008) used an optimization algorithm that found the most efficient set of sites for conservation. Specifically, an algorithm minimized the land area needed to meet conservation objectives for 58 terrestrial vertebrate species, 233 plant species, and 19 habitat types. The algorithm did not take into account habitat fragmentation or landscape integrity. Given these differences between our assessment and Wilhere et al. (2008) we expect differences in the results. Nevertheless, we also expect some congruence or correlation between our assessment and Wilhere et al. (2008).

Wilhere et al. (2008) and our assessment both exhibit the urban-to-wildlands gradient of conservation value − value steadily increases from urban areas to the foothills (Figure 2.19). Both assessments show that forests on the Kitsap and Toandos Peninsulas have relatively high value. Both assessments show that the area straddling the Pierce-Thurston County line has high value. That area has high value because of the presence of prairies and oak woodlands.

Many of the discrepancies between our assessment and Wilhere et al. (2008) can be directly attributed to the different methods used. The method of Wilhere et al. (2008) "captured" a specified amount of habitat without any information about its habitat quality or landscape integrity, while our method emphasized landscape integrity and did not have objectives specifying the amount of habitat. The resulting spatial patterns of higher relative conservation value were more diffuse in Wilhere et al. (2008) and more concentrated in our assessment.

2.4.2. Potential Improvements

Our index of conservation value could be improved several ways. First and foremost, more up-to-date and accurate species occurrence data are needed. Some of the wildlife occurrence data have not been updated in over a decade. The spatial data for prairies and oak woodlands are of unknown accuracy and do not distinguish high quality prairies and oak woodlands from highly degraded sites. A systematic survey of prairies and oak woodlands in the Puget Sound Basin that evaluates current quality and restoration potential is needed.

Second, the composition and structure of our landscape integrity index was based solely on expert judgment. We did not have the resources needed to empirically validate the index. There are many alternative formulations of landscape integrity which are also based on expert judgment (e.g., Brown and Vivas 2005, Leu et al. 2008, Theobald 2010). Further validation of our landscape integrity index could be done be comparing our index to several other independently derived indices.

Third, the landscape integrity index would be improved by developing an empirically-based statistical model that relates the composition of wildlife communities (e.g., based on similarity to "natural" wildlife communities) to various metrics of habitat loss and fragmentation. A research program attempting to develop such relationships would also investigate the multi-species habitat value of different land uses.

Further discussion of the assessments is provided in Part 5 of this report.

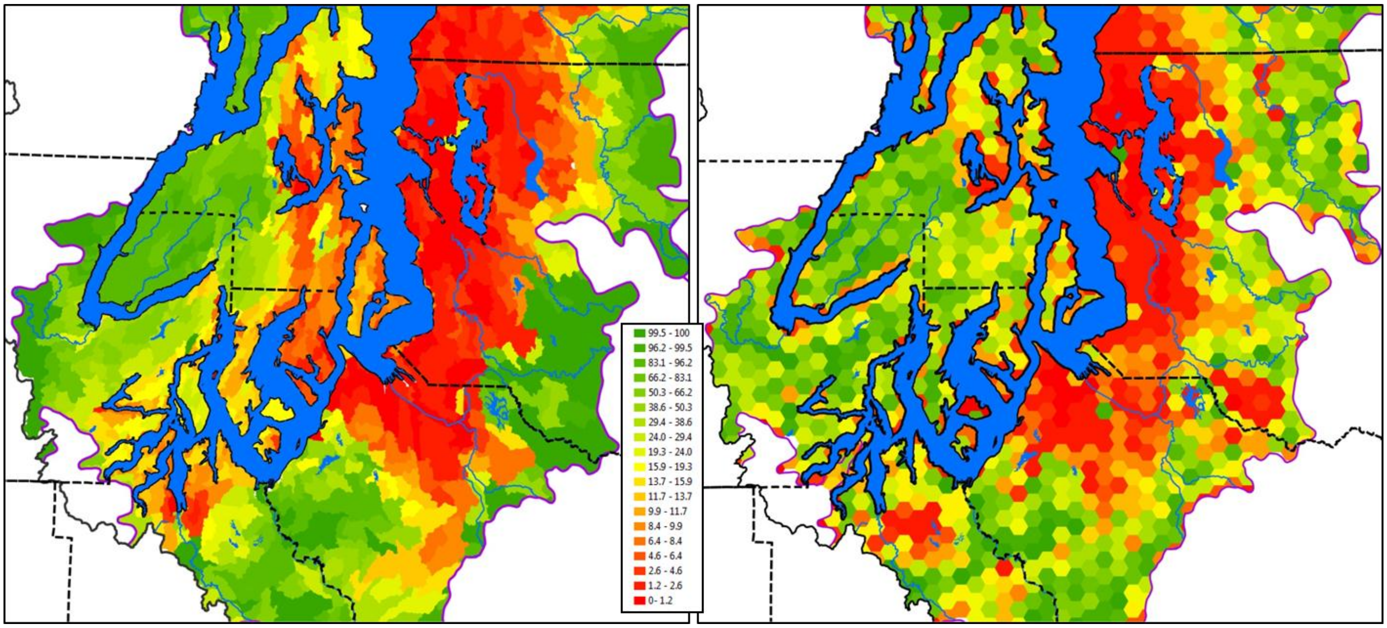


Figure 2.19. Comparison of our assessment to Wilhere et al. (2008), right and left panels, respectively. Spatial units in Willhere at al. are 750 hectare (2.9 square mile) hexagons. Purple line is boundary the Georgia Basin-Puget Trough-Willamette Valley Ecoegion. Hence, most white areas are outside the ecoregion but inside the Puget Sound Basin. Dashed lines are county boundaries. (Note: the two maps have slightly different cartographic projections).

Part 3:
Freshwater Lotic Habitats Assessment

3.1 Conceptual Model

A conceptual model is a simplified representation of a complex system that emphasizes the interrelationship of the major elements rather than the details of each element. The conceptual model describes the rationale for components and structure of the quantitative model.

3.1.1. Scientific Foundation

Three geographic properties of watersheds are fundamental to understanding lotic ecosystems: connectivity, the spatial arrangement of processes, and multiple spatial scales (Allan 2004, Wang et al. 2006). The dominant property of lotic systems is connectivity (Vannote et al. 1980, Minshall et al. 1985, Wipfli et al. 2007). A watershed is comprised of a network of connected channels that funnel materials – predominantly, water, sediment, and wood − from the watershed's headwaters down to its mouth. As materials move through the network they provide both the matter and energy for the processes that build, destroy, and rebuild aquatic habitats. Local and remote processes interact through the channel network (Figure 3.1).

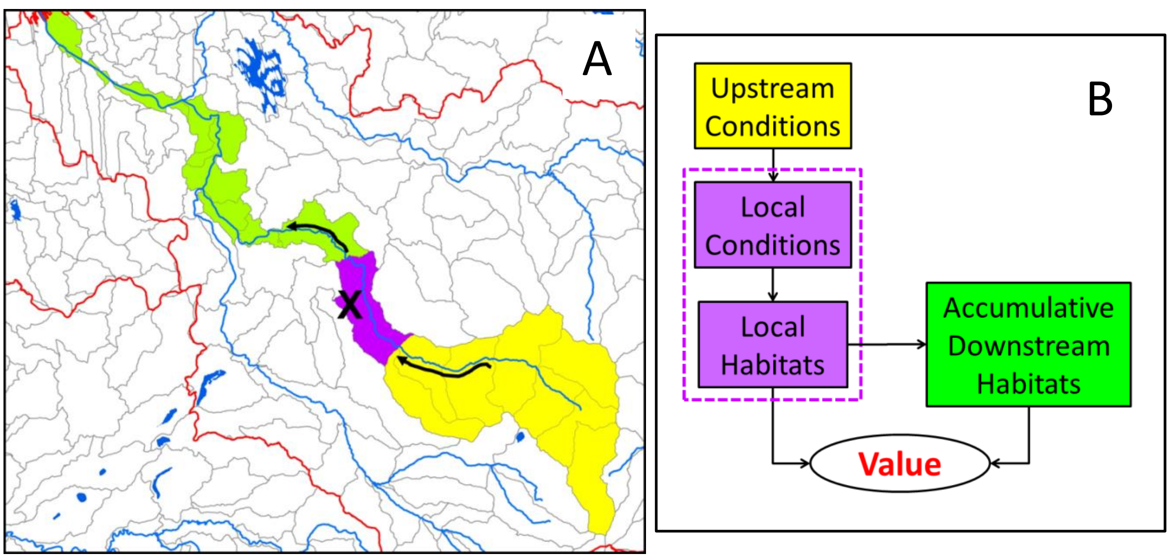


Figure 3.1. (A) Relative conservation value of a watershed is a function of what is upstream and downstream. Upstream conditions (yellow) affect habitat quality in watershed X (purple). Conditions in watershed X affect habitat quality in downstream reaches (green). Red and gray lines are WRIA and small watershed boundaries, respectively. (B) Upstream conditions affect local conditions which affect local habitats. Downstream habitats are the accumulation of local habitats. Colors correspond to those in panel A.

Processes within a watershed are arranged along two dimensions: longitudinally along the length of streams and laterally along upland hillslopes (Figure 3.2). The distance between processes along these two dimensions affects the strength of their interactions. Hence, where upland processes, in particular, anthropogenic processes, occur in relation to the stream network influence their effects on aquatic ecosystems (Gergel et al. 2002). Processes also act at various nested spatial scales. For our purposes the smallest scale is the stream reach, which is sometimes defined as a section of stream with geomorphological characteristics distinct from those of adjacent stream sections. Reach definitions often include floodplain and riparian areas adjoining the channel. The next scale is the catchment or small watershed which begins to encompass landscape influences on aquatic ecosystems (Wang at al. 2002). Larger scales entail repeated nesting of bigger and bigger drainage areas. These larger scales encompass more remote processes that may impact local processes.

Conservation value in a stream reach is affected by processes occurring upstream, and the processes in that same reach affect habitat quality downstream. Therefore, assessing the conservation value of a particular reach entails both an assessment of conditions upstream and an assessment of habitats downstream (Figure 3.1B). In other words, the value of a given stream reach is determined by: 1) habitat quality within it, which is greatly influenced by upstream conditions; and 2) the habitat quality in downstream reaches, which are influenced by the given reach's condition.

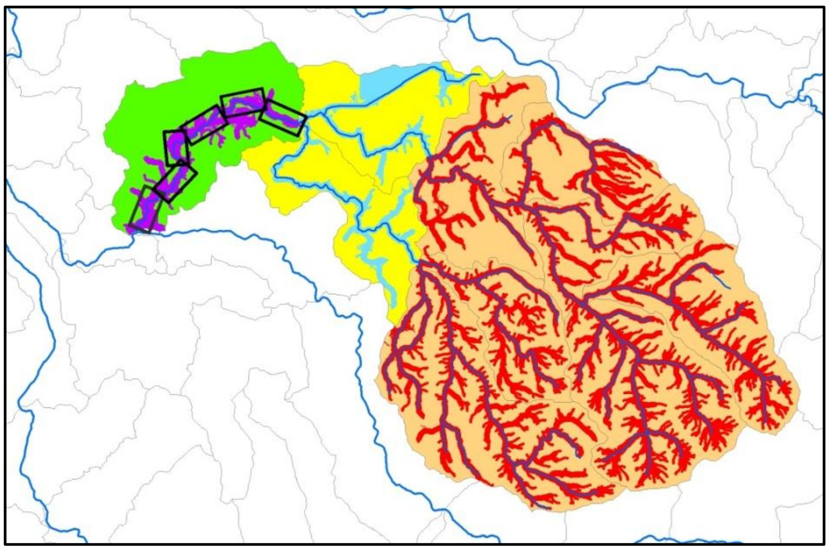


Figure 3.2. Multiple spatial scales and spatial arrangement within a watershed. The smallest scale considered in our assessment was the stream reach. Six reaches are delineated by black rectangles. The next largest scale was the small watershed or assessment unit (AU; polygons delineated by thin gray lines). The largest scale was the sub-basin (all colored AUs in the figure). Spatial arrangement has both lateral and longitudinal dimensions. The difference between riparian areas and uplands (purple versus. green, light blue versus yellow) illustrates the lateral dimension. Movement upstream from the green AU to yellow AUs to orange AUs occurs along the longitudinal dimension. Different polygon colors correspond to six distinct zones. Blue lines are rivers and streams, and only rivers and streams mapped at 1:24,000 scale are shown.

An Umbrella Species Approach

The relative value of places for fish and wildlife conservation must in some way be related to the most basic requirement of every species – habitat. In freshwater lotic ecosystems of the Puget Sound Basin the dominant vertebrate species are salmonids. We assumed that eight salmonid species and their major life-history variants – pink, chum, Chinook, coho, steelhead, rainbow trout, sockeye, kokanee, cutthroat, and bull trout[10] – could effectively serve as umbrella species for all other species that rely on

[10] Sockeye and kokanee are life history variants of *Oncorhynchus nerka*. Steelhead and rainbow trout are life history variants of *Oncorhynchus mykiss*.

lotic habitat types. An umbrella species is one whose conservation confers protection to numerous other co-occurring species (Fleishman et al. 2000). We believe this to be tenable for two reasons. First, collectively the eight species and their major life-history variants use a large proportion of every WRIA-sized[11] watershed. Those portions of a watershed where these species do not exist are very high gradient streams, headwaters, and areas above fish passage barriers. However, streams where salmonids do not exist are still important for the conservation of downstream salmonid habitats, and therefore, are covered under the umbrella species approach. Second, the egg, alevin, and juvenile life stages of salmonid species are sensitive to changes in water temperature, dissolved oxygen, and fine sediments. If these life stages are adversely affected by anthropogenic changes in a watershed, then other sensitive species may also be adversely affected. Therefore, places identified for protection or restoration of habitats for sensitive salmonid life stages will also result in the protection and restoration of habitats for non-salmonid species.

Habitat Quality

Salmonid habitats can be decomposed into intrinsic and extrinsic attributes. Intrinsic attributes consist of geomorphic or hydrological characteristics, such as channel gradient and mean annual flow, that are relatively immutable, i.e., resistant to anthropogenic changes in the watershed. Extrinsic attributes, such as water temperature, sediments, and large woody debris, are sensitive to anthropogenic changes in a watershed. Both intrinsic and extrinsic attributes influence habitat quality. Habitat quality is species-specific and is usually measured through a particular demographic response such as abundance of a life stage (i.e., number of adults or juveniles).

Models relating habitat quality to intrinsic attributes of salmonid habitats are known as *intrinsic potential models*. Intrinsic potential (IP) models are unique to each salmonid species, and perhaps even unique to salmonid populations (i.e., evolutionary significant units). IP models yield an index that quantifies the potential habitat quality of a stream reach (e.g., Burnett et al. 2007). IP models have a structure identical to that of habitat suitability models (USFWS 1981) and are usually based on expert judgment. IP models can be comprehensively applied to large regions because they use readily available, relatively high-resolution, spatially-extensive digital elevation and climate data (Busch et al. 2011). IP model results should not be mapped at a reach scale; mapping at a watershed scale best matches the accuracy of IP models (Sheer et al. 2009). IP models incorporate characteristics that are generally resistant to anthropogenic impacts, and hence, evaluate species-specific habitat potential in the absence of such impacts (Sheer et al. 2009). They attempt to estimate a reach’s potential to provide habitat and not the actual quality of habitat.

No intrinsic potential (IP) models have been developed specifically for salmon populations in the Puget Sound Basin, but IP potential models have been developed for salmon populations in other regions of the Pacific Northwest − i.e., Oregon Coast Range (Burnett et al. 2007) and the Lower Columbia River (Busch et al. 2011). The habitat relationships described by these IP models are likely to be very similar to habitat relationships for Puget Sound populations.

We were not aware of any models relating habitat quality to extrinsic habitat attributes that are general enough for a regional assessment. Progress on regional models of salmonid habitat quality has been slow to develop for three inter-related reasons. First, empirical data on extrinsic habitat attributes − e.g., water temperature, sediments, and large woody debris – are expensive to collect because (a) in-

[11] WRIA is the acronym for water resource inventory area. The Puget Sound Basin consists of 19 WRIAs that range in size from 100,000 to 1.6 million acres. The mean size is about 460,000 acres.

stream habitat measurements are physically arduous; (b) measurements cannot be performed through remote sensing techniques; and (c) some habitat attributes vary at fine spatial scales (< 0.5 km), and hence, require high sampling intensity. Consequently, modeling efforts have had to rely on datasets with small sample sizes and/or collected over a very limited geographic extent (e.g., Bartz et al. 2006), remotely sensed data such as land cover that are presumably correlated with extrinsic habitat attributes (e.g., Pess et al. 2002, Fiest et al. 2003, Steel et al. 2004, Firman et al. 2011), or "data" derived from expert opinion (e.g., Lestelle et al. 2004).

Models that relate the demographic responses of salmonid populations to extrinsic habitat attributes exist (e.g., Scheuerell et al. 2006), however, the data requirements of such models, e.g., water temperature and fine sediment measurements, are entirely impractical for a regional conservation assessment. This data problem has been addressed by developing models that relate the extrinsic habitat attributes to remotely-sensed land cover data (Bartz et al. 2006), but these models are watershed-specific (discussed further below) and cannot be generalized for regional assessments.

Some modeling efforts have ignored extrinsic habitat attributes and developed models that relate the demographic responses of salmonids to watershed-scale land cover, land use, and geology − data that can be collected via remote sensing or are readily available in spatially-extensive geographic datasets. This approach creates models that describe the relative impact of various land uses on salmonids, and therefore, such models could be useful for land use planning. While this approach has yielded watershed-specific models (e.g., Pess et al. 2002, Firman et al. 2011), it has not yet produced any regional models for salmonid habitat quality.

The second reason regional models of salmonid habitat quality have been slow to develop is that empirical data on salmonid populations − such as counts of spawners, reds, or smolts − are either unsuitable for statistical modeling (i.e., not obtained through random sampling of reaches) and/or are unavailable as geo-referenced spatial data (Ruckelshaus et al. 2002). WDFW and the National Marine Fisheries Service are currently addressing the latter problem (A. Weiss, WDFW, personal communication). These common shortcomings of salmonid data have forced modeling efforts to focus on watersheds where the data are better suited to statistical modeling techniques and/or the data have been entered into a geo-referenced spatial database. In the Puget Sound Basin, for instance, models based on empirical data relating salmon demographic responses to land use and land cover have been developed for only the Snohomish River Watershed (Pess et al. 2002, Bartz et al. 2006).

Third, because modeling efforts have had a single watershed focus, current models have a narrow geographic scope and cannot be applied to regional assessments. Models created with data from a single watershed will be regionally valid if and only if the landscape conditions within that watershed are representative of landscape conditions in all other watersheds throughout that region. However, that is unlikely to happen; consider the following. Firman et al. (2011) and Pess et al. (2002) both developed models of habitat quality for coho that used similar demographic data (counts of spawning adults) and similar landscape data such as land use, land cover, roads, and geology. The model of Pess et al. identified the amount of urban land in the watershed as a significant predictor of habitat quality but the model of Firman et al. did not. This discrepancy could be attributed to the different landscape conditions in their two study areas. The Snohomish River Watershed studied by Pess et al. is much more urbanized than the Oregon Coast Range studied by Firman et al.; the two largest cities in the Pess et al. study area have 103,000 and 60,000 people but the two largest cities in the Firman et al study area have only 16,000 and 9,100 people. Consequently, Firman et al.'s model is invalid for Snohomish River Watershed and Pess et al.'s model is invalid for the Oregon Coast Range. As for our assessment, the coho habitat model developed by Pess et al. (2002) is likely to be invalid for most of the Puget Sound Basin. The Snohomish Watershed, which is 7% urban land, is quite different from the Chambers-Clover, Cedar-Sammamish, Duwamish-Green, Deschutes, Upper Skagit, Skokomish-Dosewallips, and Lyre-Hoko watersheds which are 62, 45, 27, 21, 1, 1, and 1 percent urban land, respectively.

Because the empirical data needed to build models of habitat quality are often lacking, models derived from expert opinion are often the only practical approach. The most widely used salmonid habitat model in the Pacific Northwest, Ecosystem Diagnosis and Treatment (EDT; Lestelle et al. 2004) is based mainly on expert opinion. For the Puget Sound Basin, EDT model outputs are available for Chinook and steelhead. The EDT model includes many parameters with poorly known values, and consequently, it is prone to large error propagation and unknown levels of uncertainty (Ruckelshaus et al. 2002, McElhany et al. 2010). Because of its large number of parameters, structural complexity, and heavy dependence on expert opinion, the EDT model was severely criticized by the Salmon Recovery Science Review Panel (2000). Because of issues identified by the Salmon Recovery Science Review Panel, we did not use EDT in our assessment of relative conservation value. IP models are also based largely on expert opinion, however, IP models are structurally simple and have a small number of parameters.

Species-specific habitat models relating habitat quality to extrinsic habitat attributes were not available for the Puget Sound Basin, and we lacked the wherewithal to develop such habitat models. Consequently, we explored an approach that relied on the concept of ecological integrity. The integrity of lotic ecosystems could serve as a surrogate for the extrinsic attributes of species-specific salmonid habitats.

Aquatic Ecological Integrity

To assess relative habitat quality, we supplemented intrinsic potential with ecological integrity. *Ecological integrity* is the ability of an ecological system to support and maintain a biological community that has species composition, diversity, and functional organization comparable to those of natural habitats within a region (Parrish et al. 2003). Ecological integrity describes the degree to which an ecosystem is whole, intact, or undisturbed (Andreasen et al. 2001). Ecological integrity is much discussed in the scientific literature, but there is no generally accepted operational definition (Quigley et al. 2001). Nevertheless, ecological integrity has been assessed and mapped using spatial data such as roads, land use, land cover, housing density, or human population density that served as surrogates for ecosystem degradation (Brown and Vivas 2005, Mattson and Angermeier 2007, Theobald et al. 2010). Functional relationships between these surrogates and ecological integrity are most often formulated through expert judgment (e.g., Quigley et al. 2001, Mattson and Angermeier 2007), and rarely incorporate empirically-derived relationships between surrogates and biological responses (but see Esselman et al. 2011).

To obtain a more empirically-based and less expert-based relationship between ecological integrity and various surrogates for ecosystem degradation, we exploited published relationships between indices of biological integrity (*sensu* Karr 1991) and land use or land cover. *Biological integrity* was defined by Karr (1991) as, "the ability to support and maintain a . . . community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" − a definition apparently adopted for the definition of ecological integrity (Parrish et al. 2003). An index of biological integrity (IBI) evaluates the ecological health of rivers or streams by measuring parameters of their biological communities. The index is comprised of various metrics that quantify species richness, trophic composition, and species abundances. In the Pacific Northwest, IBIs have been developed for benthic macro-invertebrates (Fore et al. 1996), coldwater fish (Mebane et al. 2003), and coldwater fish and amphibians (Hughes et al. 2004). In addition, quantitative relationships have been developed between IBIs and various measures of anthropogenic disturbance (DeGasperi et al. 2009, Mebane et al. 2003).

Understanding the relationships between aquatic ecological integrity and land use and developing landscape-scale indicators of anthropogenic impacts on aquatic ecosystems are ongoing areas of research (Gergel et al. 2002, King et al. 2005, Burnett et al. 2006). Like models for salmonid habitat quality (explained above), relationships between aquatic ecological integrity and land use are likely to be watershed specific. While certain landscape-scale variables are widely acknowledged to be highly correlated with aquatic ecological integrity, such as percent of a watershed that is urbanized and percent of a watershed covered by impervious surface (Paul and Meyer 2001), widely-generalizable, reasonably accurate, quantitative models relating ecological integrity to land use have yet to be developed. Ecological integrity is affected by local processes, such as hillslope runoff, bank erosion, channel scouring, and wood recruitment, and the same processes occurring remotely upstream. These processes are distributed both laterally and longitudinally throughout the entire drainage area. Hence, there is a consensus among scientists that valid models of aquatic ecological integrity need to incorporate two geographic properties that are fundamental to understanding lotic ecosystems: multiple spatial scales and the spatial arrangement of processes (Allan 2004, Burnett et al. 2006).

Species Status

All species have equal inherent value, but all species do not have equal status. For instance, some species are given special status under the Endangered Species Act (ESA) because they are threatened with extinction. The listing of a species as threatened triggers actions to recover the species. For instance, under sections 7 and 9 of the ESA the habitats of listed species get special protection. Without these special habitat protections the species may not recover and decline to extinction. In effect, the habitats of species threatened with extinction are considered to be more important than the habitats of species that are not threatened with extinction. Therefore, species status could be another factor affecting the conservation value of salmonid habitats. In other words, stream reaches that contain listed salmon species would be considered more important than reaches not containing listed species.

A contrary perspective contends that healthy salmonid stocks should be the highest priority for habitat protection. This judgment stems from the belief that enhancing today's healthy populations is the most practical way to create tomorrow's sustainable fisheries. This rationale assigns a higher conservation value to watersheds containing healthy stocks than watersheds containing depressed stocks. This pragmatic perspective is especially reasonable in a rapidly urbanizing region.

3.1.2. Modeling Relative Conservation Value

The purpose of our conceptual model is to guide the construction of an index – an index of relative conservation value which will be calculated for places throughout a WRIA. These "places" are small watersheds, also known as assessment units (AUs), and hence, our model is tailored to the spatial scale of these AUs.

AUs were the spatial grain of our assessment. The spatial extent of our assessment was, in effect, each individual WRIA. That is, the freshwater habitats assessment is comprised of 18 separate assessments each corresponding to a WRIA[12]. The WRIA boundaries generally follow drainage areas, but each WRIA

[12] For the purposes of this assessment we combined WRIAs 3 and 4, the lower and upper Skagit River WRIAs, into a single WRIA.

also possesses unique patterns in geology, topography, hydrology, terrestrial vegetation, fish assemblages, and aquatic biological communities. For both scientific and political reasons, we minimized comparisons of dissimilar watersheds by doing our assessment calculations within WRIAs. Relative conservation value is relative within a WRIA, and hence, it cannot be used to compare the conservation value of AUs in different WRIAs. However, the spatial data and assessment methods were consistent across WRIAs, and therefore, the assessment can be used to discern patterns in relative conservation value among WRIAs.

The principal challenge we faced in developing the index were the limitations imposed by the currently available spatial data. Occurrence data for native freshwater animal species collected by WDFW and other agencies focus almost entirely on harvested species, and consequently, we have reasonably accurate data across the entire Puget Sound Basin for only salmonid species. The shortcomings of the available spatial data led to an assumption that the eight salmonid species and their major life-history variants could effectively serve as umbrella species for all other species that rely on lotic habitat types. Consequently, salmonid species richness[13] and the amount and quality of salmonid habitats are major influences within the index.

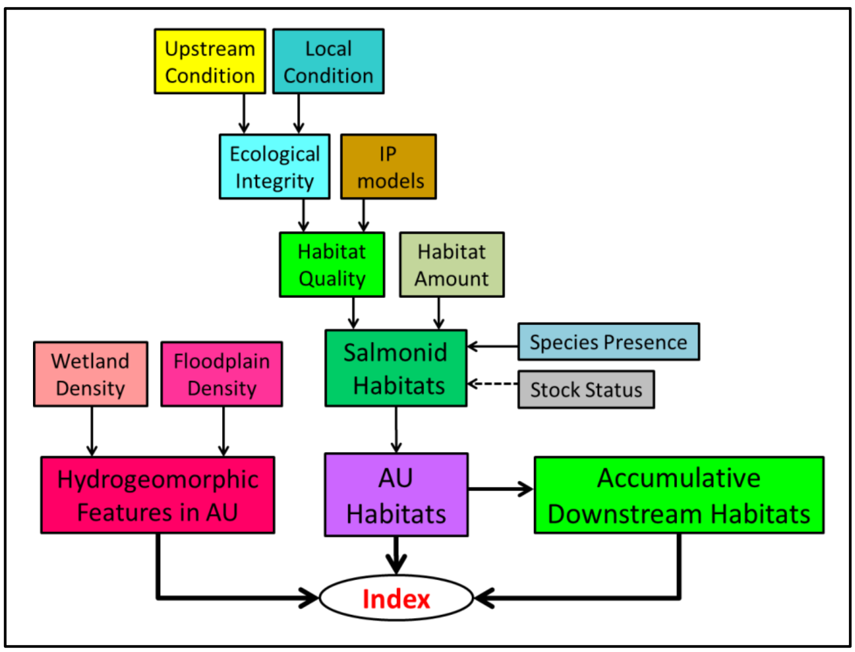


Figure 3.3. Components of the relative conservation value index. The three main components are hydrogeomorphic features in the AU, salmonid habitats in the AU, and accumulative downstream salmonid habitats. The dotted line from stock status indicates that this component was included only in a secondary analysis done to examine the effects of stock status on relative conservation value. IP = intrinsic potential.

The components of the index are organized as 5 tiers (Figure 3.3). The bottom tier has three components: hydrogeomorphic features, local salmonid habitats, and accumulative downstream habitats. Hydrogeomorphic features refer to the density of wetlands and undeveloped floodplains,

[13] Species richness means the number of species at a location.

which are landscape-scale features crucial to ecological processes that create and maintain lotic habitats. Local habitats are the salmonid habitats inside an AU, and accumulative downstream habitats are the salmonid habitats outside and downstream from an AU.

On the next tier the main component is salmonid habitats. Within this component separate calculations are done for eight salmonid species. The relative value of each salmonid habitat is a primarily a function of habitat quality and habitat amount, but it is also influenced by species' presence, and optionally, the species' status. Salmonid *habitat quality* is a function of intrinsic potential and aquatic ecological integrity, which address the intrinsic and extrinsic attributes of freshwater salmonid habitats, respectively. Ecological integrity of an AU depends on conditions within that AU and on conditions upstream of the AU. Because of the spatial scale of the assessment and the available data, AU and upstream conditions must be based on land use and land cover.

We assume the habitat occupied by salmon is more valuable than unoccupied habitat. Hence, salmonid *habitat value* combines reach habitat quality with species occurrence information for each reach. Our fish occurrence data have three categories of presence – documented, presumed, and potential (Appendix C) – that reflect the level of certainty regarding species occurrences. Certainty of species occurrence affects habitat value. Species status could be another factor affecting habitat value, however, because incorporating species status incorporates, in effect, a legislative policy, we did not include status as a factor affecting conservation values. However, we were curious as to how species status would affect the results. Consequently, we did a secondary analysis that included ESA status and the Salmonid Stock Inventory status (WDFW and WWTIT 2002).

There are several ways an AU can be highly valuable for the conservation of salmonid habitats – the AU contains exceptionally high quality habitat for only one species; contains large amounts of habitat for many species, regardless of habitat quality; contains some intermediate amounts of high quality habitats for some species, or contains large amounts of moderate quality habitats for some species, etc. In other words, conservation value is a function of habitat quantity, quality habitat, and species richness. Our index incorporated all three aspects of conservation value.

Another way that an AU can be valuable for the conservation of freshwater lotic habitats is its potential impact on downstream habitats. AUs that could potentially impact large amounts of high quality habitat should be protected in order to avoid or minimize adverse impacts on those downstream habitats. For each AU, our index quantified the value of downstream habitats.

There are two basic perspectives on modeling the relative conservation value of places, and they reflect a quantity versus quality dichotomy. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's single most significant contribution, i.e., the quality a place contributes. These two perspectives can result in different rankings of places. For example, the former perspective would value a place with high species richness over a place with high species rarity, while the latter would value rarity over richness. Neither perspective should be ignored, so we examined relative conservation value both ways.

3.2 Methods

A detailed explanation of the methods is given in Appendix B.

Our index of relative conservation value was based on expert judgment. The use of expert judgment as a substitute for empirical information has been criticized (Ruckelshaus et al. 2002), but we had no other practical alternative. Judgments were made regarding the components of the index, how to assemble them, and the relative influence of each component.

3.2.1. Spatial Framework

The assessment units (AUs) are the same AUs used by the Department of Ecology for their assessments of water resources (Stanley et al. 2010). AUs were derived from reach-scale catchments delineated by the Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP; NWIFC 2009). Ecology and WDFW believed that the water resources and habitats assessments could not be accurate at this catchment scale due to the resolution of some spatial data layers (i.e. 1:24,000 and smaller). We thought AUs on the order of 1 to 10 square miles were more reasonable. Consequently, the SSHIAP catchments were aggregated into larger analysis units. Two-thousand nine-hundred forty AUs with a mean size 4.7 square miles were created.

Some components of our index needed specific information on individual river and stream reaches: length, channel gradient, active channel width, valley floor width, and mean annual flow. No currently available GIS data layers provided such information, so we contracted with M2 Environmental Services to create one. The resulting "NetTrace" channel network for the entire Puget Sound Basin had 706,744 stream reaches with an average length of 117 meters (Table B2).

3.2.2. The Index

The main components of the index are 1) the density of wetlands and undeveloped floodplains, 2) local salmonid habitats, and 3) accumulative downstream habitats (Figure 3.3).

Hydrogeomorphic Features

Spatial data for wetlands was obtained from Department of Ecology (Stanley et al. 2011). We refined the wetland data layer by overlaying it with a land cover/land use data layer (C-CAP 2008) and removing wetlands co-incident with urban or agricultural land uses. The area of functional floodplains was calculated by removing areas that were co-incident with "developed" land uses in C-CAP.

With respect to hydrogeomorphic features, an AU has high value if it has a high density of wetlands and undeveloped floodplains or contains a high proportion of a WRIA's wetlands and undeveloped floodplains. Hence, an AU's relative value for hydrogeomorphic features was calculated two ways, and the hydrogeomorphic feature component of the index equaled the larger of the two resulting values.

AU Habitats

Local or AU habitat value was a function of habitat quality, habitat amount, and fish presence category. WDFW's FishDist database (Figure B4) was the source of all spatial data on the presence of salmonids in rivers and streams. FishDist data for 10 salmonid species and life-history variants – Chinook, coho, pink, chum, sockeye, kokanee, steelhead, rainbow trout, cutthroat, and bull trout – were transferred to reaches in the NetTrace channel network. For the purposes of this analysis, we equated presumed presence with documented presence but assigned lesser value to water bodies where a salmonid species had potential presence (Table B6). To simplify the analysis we lumped kokanee with sockeye, and where steelhead and rainbow trout co-occur we lumped them together also.

Habitat quality was the weighted geometric mean of intrinsic potential and ecological integrity (Figure B3). We currently have IP models for steelhead, coho, and Chinook (Figure B6). The steelhead model was also applied to rainbow trout. For those salmonid species that lack an IP model, intrinsic potential was set equal to 1, and consequently, habitat quality was only a function of ecological integrity.

None of IP models we utilized for this assessment were developed specifically for Puget Sound salmon populations, and IP models specifically for Puget Sound salmonid populations are likely to be different. However, we believed that the available models were adequate for our purpose; namely, to calculate watershed-scale estimates of relative conservation value and make valid distinctions among AUs.

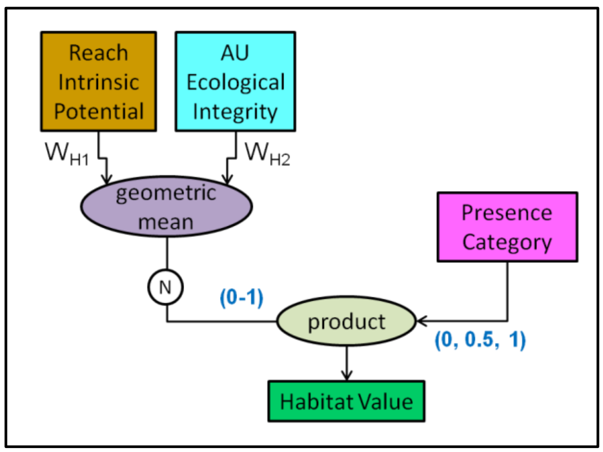


Figure 3.4. Model for salmonid habitat value. The weighted geometric mean of intrinsic potential (IP) and aquatic ecological integrity equals habitat quality. Values assigned to weights, W_x, given in Table B5. In the models for salmonid species for which we lack an IP model, intrinsic potential equals 1. N stands for normalization which is done within WRIAs. An alternative model includes species and stock status (see Figure B5)

To develop our index of aquatic ecological integrity we utilized two studies that found significant relationships between indices of biological integrity (IBIs) and the proportion of a watershed covered by certain land covers or land uses: Mebane et al. (2003) and DeGasperi et al. (2009). Both DeGasperi et al. (2009) and Mebane et al. (2003) performed straight line regressions on their data. We conducted our own analyses and found for both sets of data that better fits were obtained with power functions. We used these new relationships in our calculation of ecological integrity. Our index of aquatic ecological integrity is ultimately based on land cover. The three “predictor” variables for ecological integrity were percent of a watershed covered by impervious surface; percent of a watershed not covered by forest, wetlands, or natural vegetation; and percent of a watershed covered by human disturbances (e.g., urban, residential and agricultural).

The ecological integrity of aquatic habitats is governed by processes occurring both locally and remotely. Hence, we applied the ecological integrity functions to six zones that divided a drainage area along both lateral and longitudinal dimensions (Figure 3.2). The two lateral zones were floodplains/riparian areas and uplands. The three longitudinal zones were 1) the focal AU, 2) AUs immediately upstream of the focal AU, and 3) all other AUs in the upstream drainage area of the focal AU. Ecological integrity index values calculated for these six zones were combined with a weighted arithmetic mean to yield a composite ecological integrity (CEI) index for each AU. The equation for CEI was of the form:

$$CEI = \sum_{i=1}^{6} I_i \; P_i \; W_{L1}(longitudnal_i)\, W_{L2}(lateral_i) \tag{3.1}$$

where i denotes one of the six zones in Figure 3.2; I_i is the aquatic ecological integrity index for zone i; P_i is the proportion of an AU's entire contributing drainage basin covered by zone i; and W_{L1} and W_{L2} are weights reflecting the relative influence of each zone on ecological integrity. W_{L1} and W_{L2} are functions – formulated through professional judgment – of the longitudinal and lateral positions of zone i within the drainage basin.

Habitat value equals habitat quality combined with species presence category (and optionally the species and stock status; Figure B5). Habitat value is calculated for each species present in a reach. Hence, up to eight values per reach must be summarized into a single value. We derived two separate indices that combine the eight values (Figure B9): the maximum habitat value per reach and the sum of the habitat values times the habitat amount (i.e., reach length). Habitat value times habitat amount equals habitat units. A reach contributes the largest amount of habitat units when it is long and has high habitat value, but exceptionally long reaches with low habitat value and short reaches with exceptionally high habitat value can also contribute a large amount of habitat units.

Using only one of the two metrics would fail to identify many high value reaches. Maximum habitat value identifies reaches that contain exceptionally high quality habitat for only one species, while the sum of habitat units identifies reaches with a large amount of habitat for many species. Hence, the reach habitats index (RHI) is the maximum of the two metrics (Figure B9). RHI is used in the calculation of accumulative downstream habitats

AUs are small watersheds. Hence, the reach-scale habitat values and habitat units must be combined to yield a watershed-scale index. The watershed habitats index (WHI) for an AU equals the maximum of either the sum of habitat units for all stream reaches in the AU or the sum of habitat units for reaches in that AU with a maximum habitat value greater than the 80th percentile habitat value for the WRIA where the AU is located (Figure B10). In other words, WHI assigns a high value to AUs that either have a relatively large amount of habitat units or have a relatively large amount of high value habitat.
Before applying the maximum function, the two components of WHI were divided by AU area to yield a habitat unit density and normalized by their respective maximum values within the WRIA.

Accumulative Downstream Habitats

The calculation of the accumulative downstream habitats component of an AU's relative conservation value was done in two steps. First, for each reach, RHIs for all downstream reaches were summed (Figure 3.5). M2 Environmental Services created a computer program that performed this operation. Second, the reach-level accumulative downstream habitats values were averaged within each AU.

The Indices of Relative Conservation Value

We have three components with which to calculate an index: hydrogeomorphic features, local habitats (i.e., WHI), and accumulative downstream habitats (Figure 3.6). We calculated two indices: an average of the three components and their maximum value. For the purposes of combining these three components, the values were ranked relative to other AUs in their WRIA, and the ranks were normalized to yield a range of scores from 0 to 1.

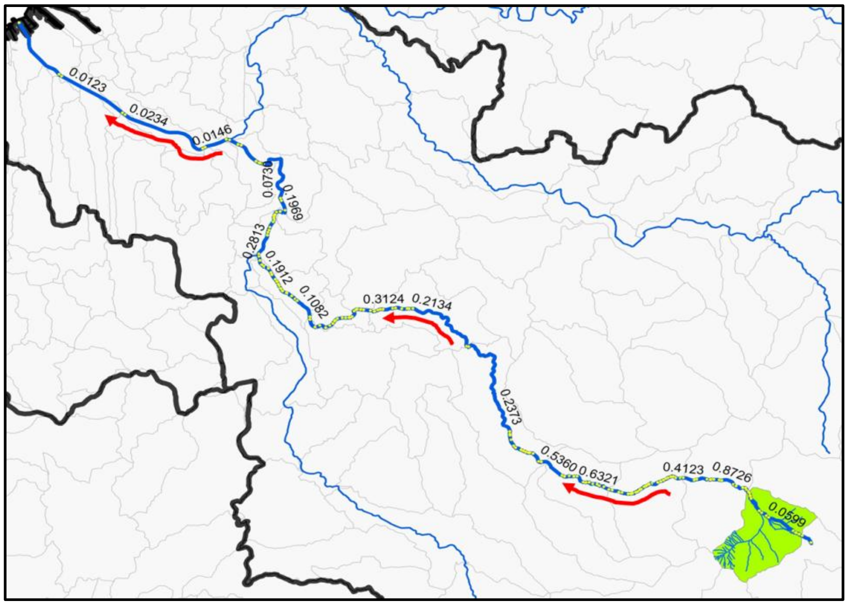


Figure 3.5. The accumulative downstream habitats (ADH) component of the index of relative conservation value. ADH for the green AU is the sum of RHI values downstream of the AU. Yellow dots mark breaks between adjacent stream reaches, and numbers are hypothetical RHI values for each reach. Gray lines area AU boundaries, thick black lines are WRIA boundaries, and blue lines are rivers.

3.2.3. Sensitivity and Uncertainty Analyses

Sensitivity analysis was done by calculating the index for all AUs with the parameter values shown in Tables B5 amd B6, recalculating the index after altering a single parameter by a small amount (e.g., 5%), and applying equations 1.2 and 1.3. The process was repeated for each parameter. Another sensitivity analysis examined how the salmonid habitat index changed in response to changes in model structure. We examined four major changes to the model: removing ecological integrity, removing intrinsic potential models, removing both ecological integrity and intrinsic potential (i.e., habitat quality), and including species status.

Uncertainty analysis was done by assigning a uniform probability distribution to each of 11 parameters (Table B5). The distributions spanned the range of reasonable values for each parameter. Over 2,000 iterations a parameter value was randomly selected from each distribution and the index was recalculated for each AU.

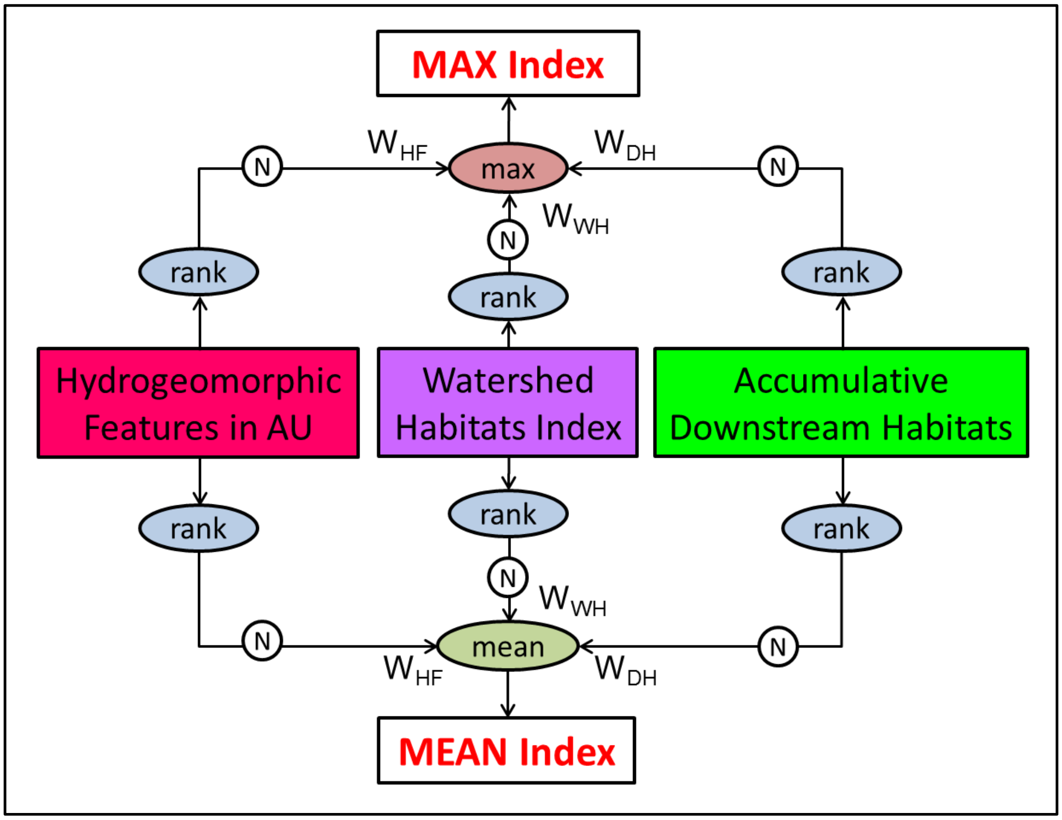


Figure 3.6. Two indices generated from the three components of relative conservation value: the mean index and max index. Ranks are normalized, so both indices range from 0 to 1. W_x are weights, which were all set to 1 for this analysis.

3.3 Results

To understand the relative conservation value indices we must examine their main components: hydrogeomorphic features, local habitats, and accumulative downstream habitats.

3.3.1. Hydrogeomorphic Features in AU

Wetlands and floodplains have their highest densities in the Puget Trough lowlands outside of urban areas (Figure 3.4). The statistical distribution of densities was highly skewed to the right (Figure 3.5). The mean wetland and floodplain density (expressed as percent of AU area) in the Puget Sound Basin was 14%, but the mean value by WRIA ranged from 9% in WRIAs 12 and 17 (Chambers-Clover and Quilcene-Snow) to approximately 22% in WRIAs 1 and 11 (Nooksack and Nisqually). AUs with high densities typically contained large rivers with mostly undeveloped floodplains. The densities of hydrogeomorphic features were converted to normalized ranks within WRIAs, hence, the statistical distribution of values was roughly uniform (Figure 3.10C).

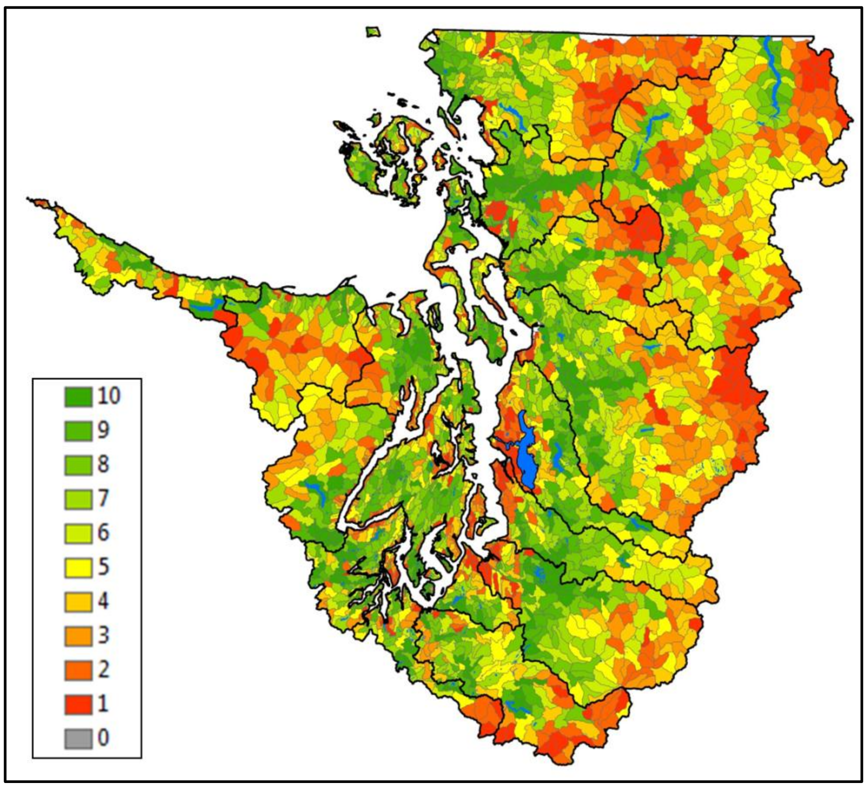


Figure 3.4. Relative density of extant wetlands and undeveloped floodplains. The values 1 to 10 represent 10 deciles for the frequency distribution of wetland and floodplain densities in AUs. That is, analysis units (AUs) in top 10% of AUs in a WRIA are in the 10th decile (darkest green), and AUs in the bottom 10 % of AUs in a WRIA are in the 1st decile (darkest red). Deciles were calculated separately for each WRIA.

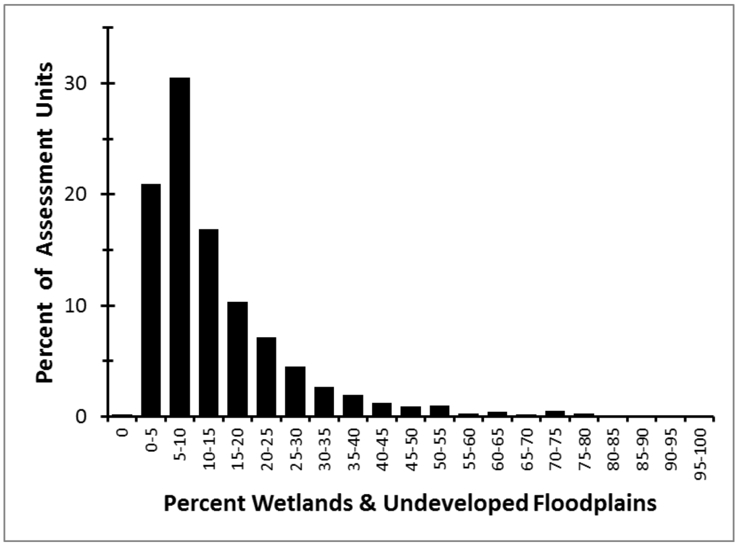


Figure 3.5. Distribution for the densities of extant wetlands and undeveloped floodplains in assessment units (AUs). Density is expressed as percent of AU area. There are 2940 AUs. Median value was 9.7% and mean value was 13.9%.

3.3.2. Local Habitats

Recall that WHI had two components: (1) the sum of all habitat units of all reaches in an AU (sumHU), and (2) the sum of habitat units of all reaches in the AU with maximum habitat value greater than the 80th percentile habitat value for the WRIA (sumHU^{80}).

Thirty-four percent of AUs had zero value for sumHU because no salmonids are documented, presumed, or have the potential to inhabit them, according to our data. These 34 percent of AUs cover 15 percent of the Puget Sound Basin's land area. An additional 14% of AUs had non-zero sumHU but had zero sumHU^{80} because none of their habitats exceeded their WRIA's 80th percentile habitat value. Because the sumHU and sumHU^{80} indices are normalized ranks, the distributions of their nonzero values are uniform with mean and medians of approximately 0.5.

The Pearson product-moment correlation between sumHU and sumHU^{80} was 0.88. This correlation is high but not so high as to suggest that the two indices are redundant. The mean absolute difference between non-zero values of sumHU and sumHU^{80} was 0.15 and 25% of the differences were greater than 0.22 which indicate the two indices are conveying different information. WHI is the maximum of sumHU and sumHU^{80}. SumHU was bigger than sumHU^{80} for 54% of AUs with non-zero WHI.

Values for sumHU were greatest in valley bottoms of large rivers (Figure 3.6). This mostly was a result of greater salmonid species richness but also was determined in part by the intrinsic potential models for Chinook and coho which specify that low gradient streams have the greatest potential. Many AUs with large sumHU^{80} values were located higher in a WRIA where ecological integrity tended to be higher. Very rarely were the highest value habitats located in urban or agricultural areas; a result largely driven by the ecological integrity index. Zero values for sumHU^{80} were located in urban and agricultural areas and in high-elevation AUs with relatively little salmonid habitat.

A major component of the reach habitat index was ecological integrity. The spatial pattern of ecological integrity conformed to our expectations (Figure 3.7). Ecological integrity was highest in the Cascades and Olympic Mountains and lowest in the Puget Trough lowlands near urban areas. The statistical distribution of values was bimodal (Figure 3.8) with one mode between 95 and 100 corresponding to AUs in the mountains. The mean value in the entire Puget Sound Basin was 53, but the mean value by WRIA ranged from 26 in WRIA 12 (Chambers-Clover) to 71 in WRIA 16 (Skokomish-Dosewallips). Unlike other components of the freshwater habitats assessment, the ecological integrity index was not normalized within WRIAs.

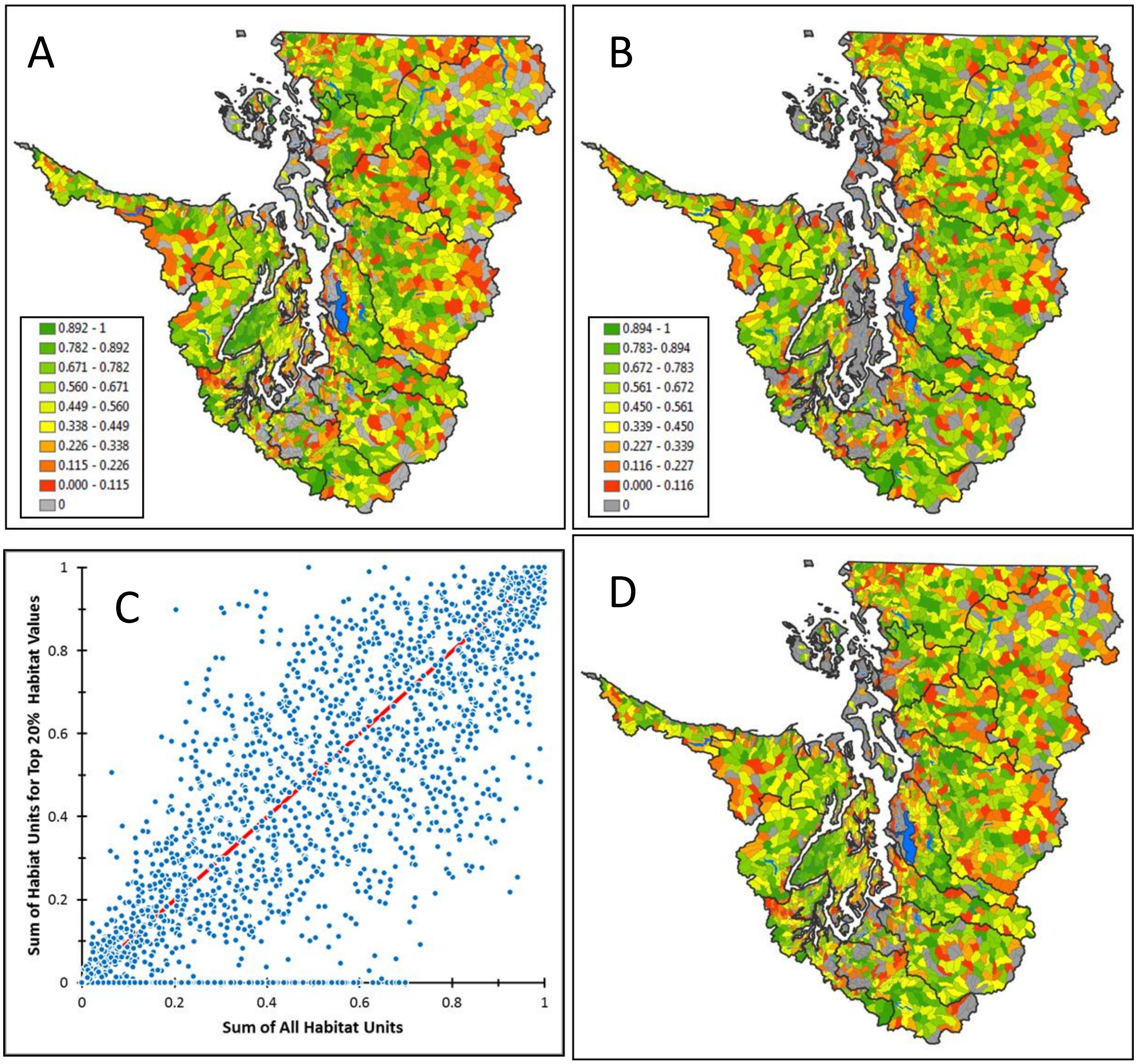


Figure 3.6. Two components of the watershed habitats index (WHI) and the resulting WHI. (A)The sum of all habitat units of all reaches within each AU. (B) The sum of habitat units of reaches in the AU with maximum habitat value greater than the 80th percentile habitat value for the WRIA. Gray AUs either do not have salmonids present (according to our data) or do not contain reaches with habitat value greater than the 80th percentile in WRIA. Both indices are normalized ranks, where ranks are among AUs within a WRIA. (C) Plot of index in Map B versus the index in Map A. (D) WHI created by taking the maximum value for each AU in Maps A and B and then renormalizing values within a WRIA.

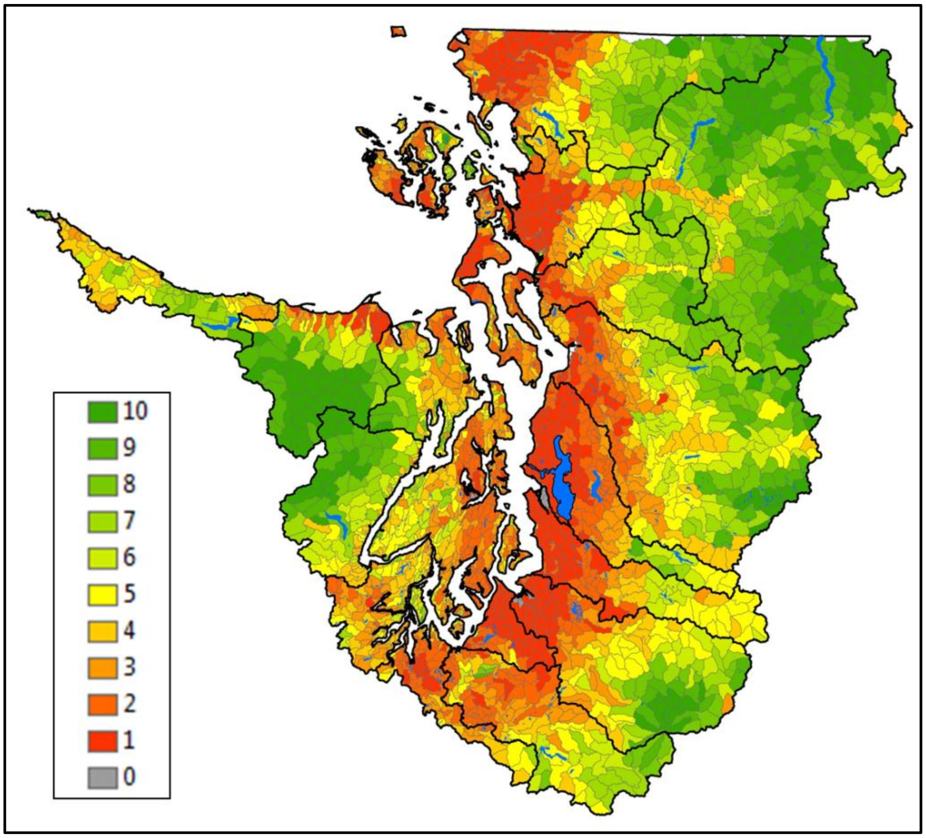


Figure 3.7. Ecological integrity calculated for each AU. Results are presented as quantiles (i.e., deciles) by area. That is, approximately 10 % of the Puget Sound Basin's land area is in the 10th decile (top 10%, darkest green), 10% of the Basin's area is in the 9th decile, and so on.

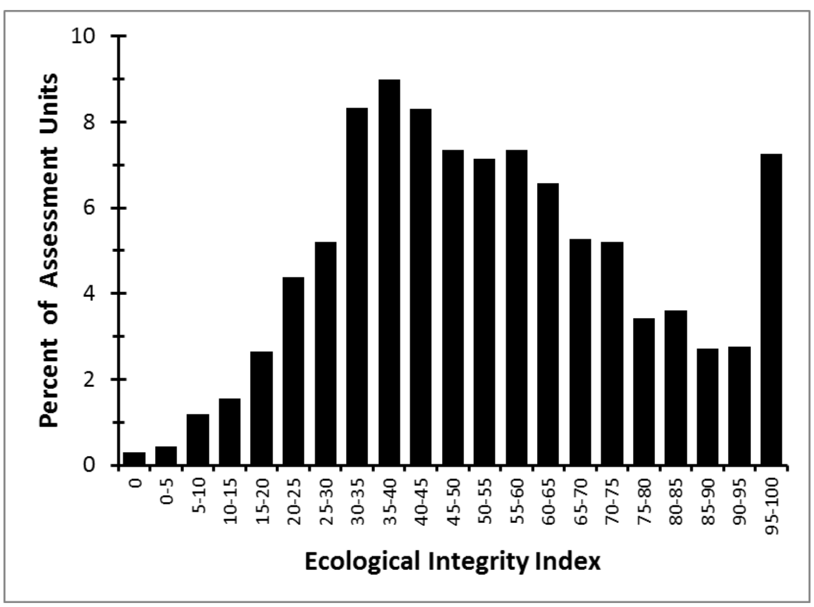


Figure 3.8. Distribution of ecological integrity index values calculated for assessment units (AUs). There are 2940 AUs. Median value was 51 and mean value was 53.

3.3.3. Accumulative Downstream Habitats

The spatial pattern of accumulative downstream habitats conformed to our expectations (Figure 3.9). The headwaters of large rivers affect the largest amount of lotic habitats. As one progresses from a river mouth toward the Cascade Crest, the accumulative effects of a stream reach on downstream habitats increases. Twenty-eight percent of AUs had a zero value for of accumulative downstream habitats.

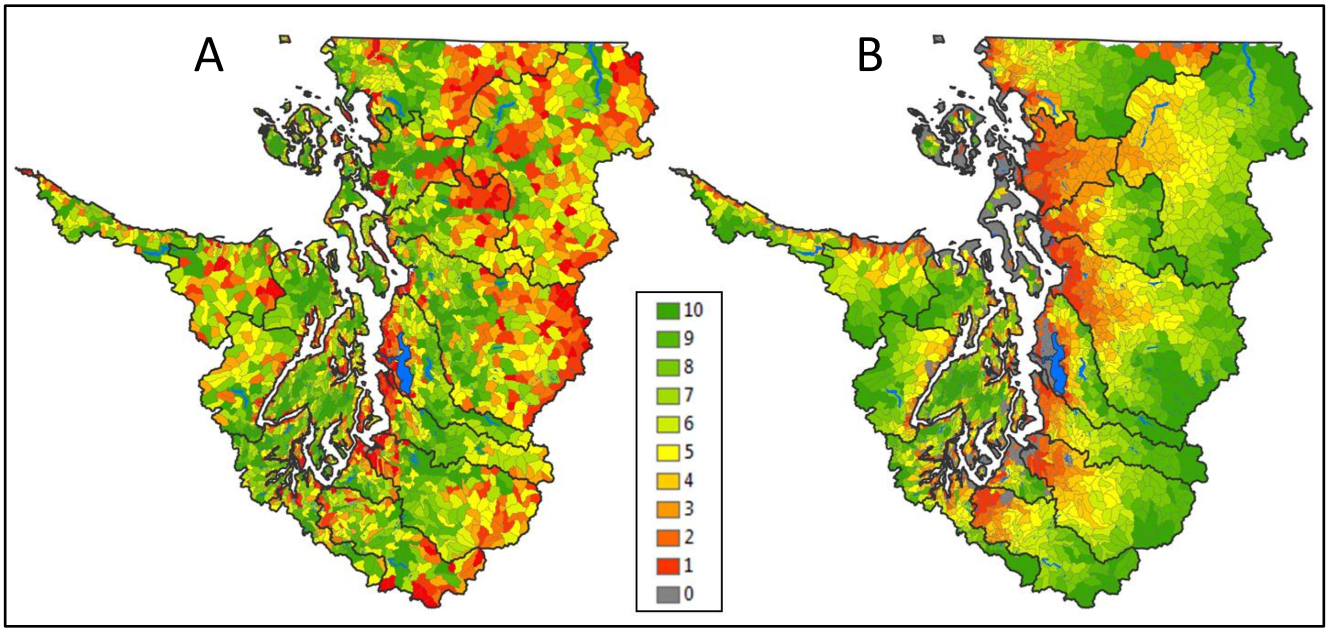


Figure 3.9. Comparison of hydrogeomorphic features index (A) with accumulative downstream habitats index (B). Results presented as deciles. For instance, approximately 10% of each WRIA's AUs are in the 1st decile (bottom 10%, darkest red) and 10% of each WRIA's AUs are in the 10th decile (top 10%, darkest green). Quantiles calculated separately for each WRIA.

The spatial pattern of accumulative downstream habitats is nearly the opposite the pattern for wetlands and undeveloped floodplains. AUs with a high relative density of wetlands and floodplains tend to be located lower in a WRIA and consequently have lower relative value for accumulative downstream habitats, and AUs with high relative value for accumulative downstream habitats tend to be located in foothills or mountains and consequently have a lower density of wetlands and floodplains.

3.3.4. Relative Conservation Value

Recall that we wish to convey two perspectives regarding conservation value. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's single most significant contribution, i.e., the quality a place contributes.

The first perspective is served by the average of components. The average shows that multiple functions are most likely to be performed in the lowlands (Figure 3.10). This is where high density of wetland and floodplains and high relative values for local habitats are likely to co-occur. AUs that perform the fewest functions are those at higher elevations. Most of those AUs obtain high relative values for accumulative downstream habitats only. The distribution of the average values had a mean and a median equal to 0.40, and the distribution's shape was relatively uniform for values between 0 and 0.70 (Figure 3.10D). No AUs had an average value over 0.95, but about 1% of AUs had a high score (≥0.8) for all three components.

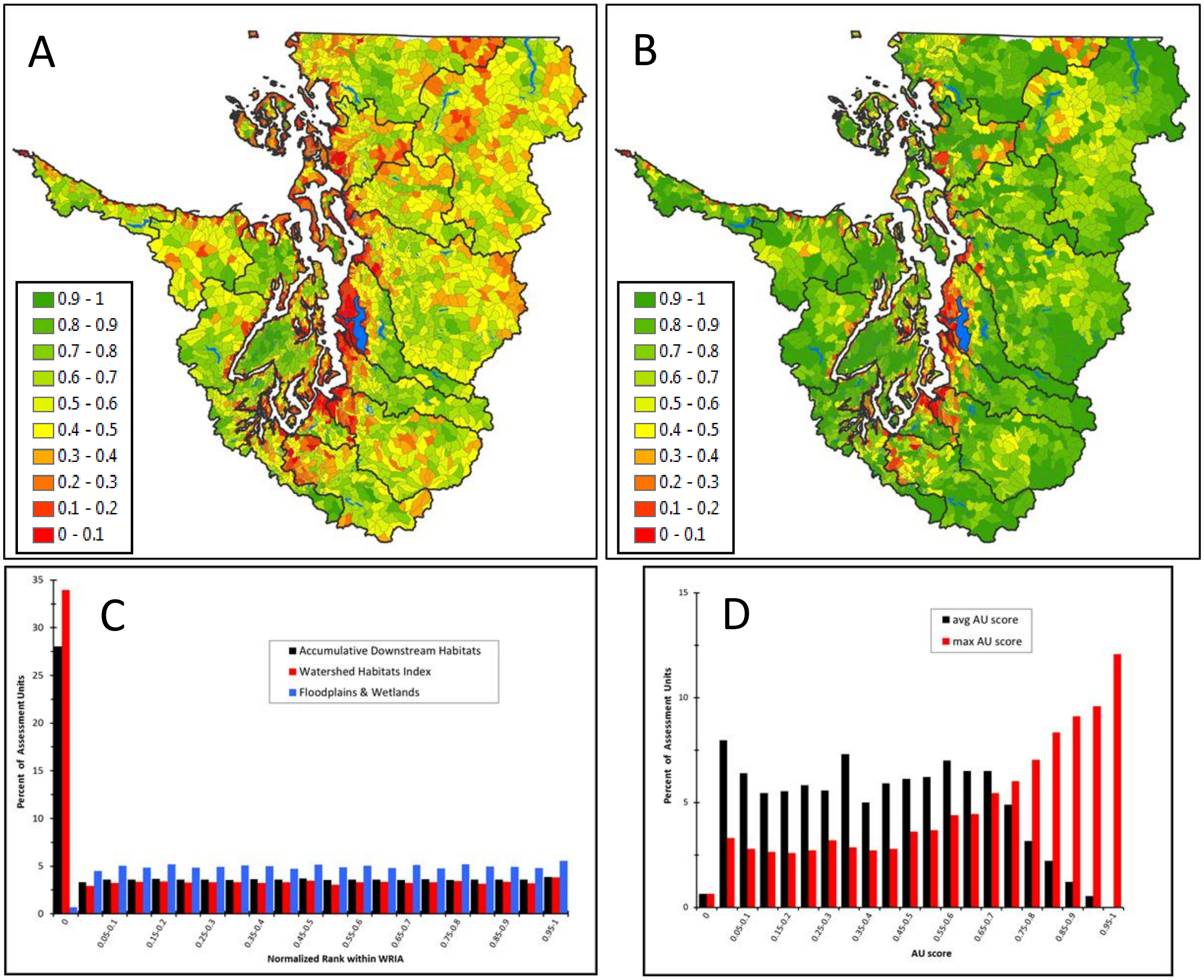


Figure 3.10. Two perspectives on relative conservation value. (A) Average of the three main components of relative value: hydrogeomorphic features, watershed habitats index, and accumulative downstream habitats. (B) Maximum value of the three components for each AU. (C) Index distributions for the three components which are normalized ranks; AUs were ranked within WRIAs. (D) Distribution of values in Maps A and B.

The second perspective is served by the maximum of components. Seventy percent of AUs had a score for at least one of the components greater than 0.5. The median value of the maximum of components was 0.72, which means half of all AUs attained a score of 0.72 or greater for at least one of the three components. Maximum scores greater than 0.90 were attained by 22 percent of AUs, which means about one-fifth of AUs were in the top 10% of AUs in their WRIA for at least one of the components (recall that the scores are normalized ranks). The map of maximum scores (Figure 3.10B) shows that, with the exception for urban areas, scores above 0.5 are distributed almost evenly throughout the Puget

Sound Basin. Hydrogeomorphic features, downstream accumulative habitats, and WHI were the maximum component for 40%, 31%, and 29%, respectively, of AUs with nonzero WHI.

Table 3.1. Statistical summary of two perspectives on relative conservation value.

Statistic	Average of 3 Components	Maximum of 3 Components
mean	0.40	0.64
standard deviation	0.24	0.29
minimum	0	0
1st quartile	0.19	0.43
median	0.40	0.72
3rd quartile	0.60	0.88
maximum	0.93	1

3.3.5. Sensitivity and Uncertainty Analyses

The weight parameters with the biggest influence on the index were the two weights determining the relative influence of salmon species with IP models and species without IP models. These two weights have the largest sensitivity because among those parameters evaluated for sensitivity, they are in an equation which is furthest along in the calculation of WHI (in the bottom in Figure 3.3). In the index equation these two parameters were equal. Changing either of these parameters by 1% results in a nearly 1% change in WHI averaged across all AUs (Figure 3.11).

The sensitivity analysis showed that parameters affecting the most AUs have the greatest influence on the results. This is a common finding of the sensitivity analyses done for the terrestrial, freshwater, and marine shoreline assessments. For example, amongst the three parameters involved with the type of species presence, potential presence had a much smaller influence than documented and presumed presence because potential presence was much less common in the species occurrence database.

The mean sensitivity of WHI to change in parameters for individual AUs was not related to WHI (Figure 3.12). Mean absolute elasticity was showed no trend as WHI increased. For 95% of AUs mean absolute elasticity was 0.06%. That is, for 95% of AUs a 1% change in parameter value results, on average, in a less than 0.1% change in WHI. However, some AUs were very sensitive to changes in parameters – the three largest values for mean absolute elasticity were 5.1, 4.7, and 4.7 percent.

Another sensitivity analysis examined how WHI changed in response to changes in model structure. Adding species status to the index results in 22% of AUs with nonzero WHI changing their quantized WHI, but the change was only 1 decile for 20% of AUs (Table 3.2). Removing ecological integrity from the index results in 68% of AUs with nonzero WHI changing their quantized WHI, however, 32% changed by only 1 decile. Removing the ecological integrity index had a bigger influence on WHI than removing the intrinsic potential models. Removing components from WHI does not result in dramatic changes in quantized WHI because changes to the index affect all AUs similarly.

Uncertainty was small for WHI values near 1. WHI scores greater than 0.9 had a mean coefficient of variation of 2%. For AUs with non-zero WHI, the median width of the 90% confidence interval for WHI was 0.17. In other words, given our expert uncertainty about model parameters, the WHI value for most AUs is very likely within ±0.085 of the calculated value. However, for 10% of non-zero WHI scores the width of the confidence interval was 0.42. In other words, for 197 AUs a WHI calculated to be 0.7, for instance, has a good chance of being as high as 0.91 or as low as 0.49.

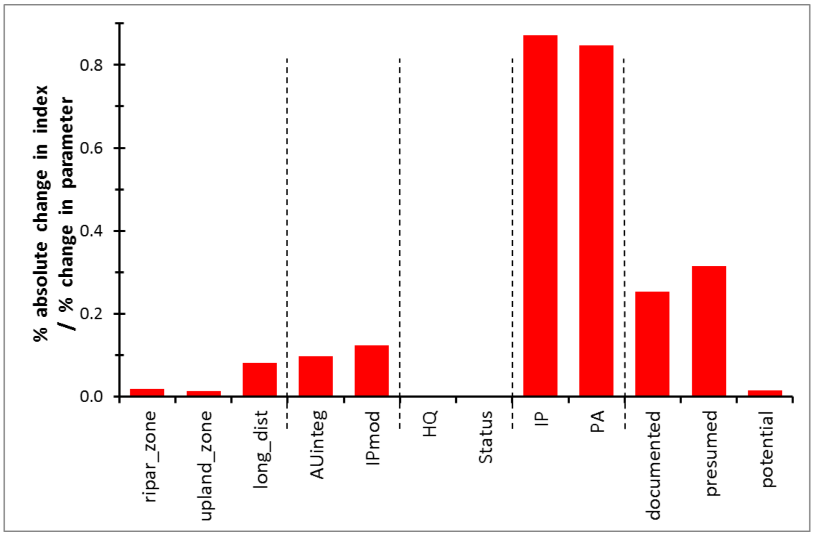


Figure 3.11. Results of sensitivity analysis: elasticity of WHI to changes in model parameters. Vertical dashed lines indicate parameters in the same equation. Habitat quality (HQ) and species status have zero elasticity because status was not included in the index, and HQ and status are the only two variables in a particular equation. IP = salmonid species with an intrinsic potential models; PA = salmonid species with no IP model (i.e., presence/absence data only). See Tables B5 and B6 for description of weight parameters.

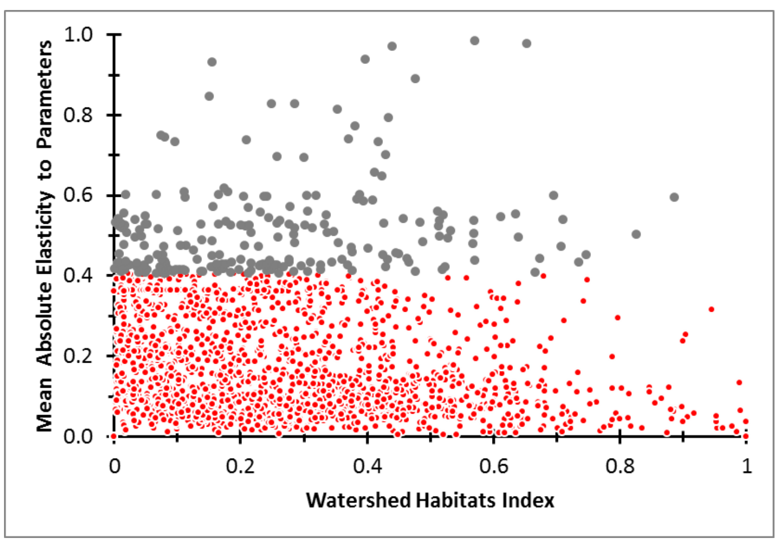


Figure 3.12. Results of sensitivity analysis: mean absolute elasticity of WHI per AU to changes in model parameters. Gray points are AUs among the 10% of AUs with largest mean absolute elasticity. Red points are remaining 90% of AUs. Largest mean absolute elasticity was 5.1%.

Table 3.2. Percent of AUs with change in WHI decile in response to changes in WHI structure. Only AUs with non-zero WHI were tabulated. Number of AUs with non-zero WHI equals 1942.

	Change in Index Structure			
Change in WHI Decile	Include Species Status	Remove Ecological Integrity	Remove Intrinsic Potential	Remove Habitat Quality
-8	0.0	0.0	0.0	0.0
-7	0.1	0.0	0.0	0.0
-6	0.1	0.1	0.1	0.1
-5	0.1	0.1	0.3	0.2
-4	0.1	1.6	0.7	1.8
-3	0.1	6.0	0.8	6.1
-2	0.3	11.6	3.8	11.2
-1	10.6	19.3	14.8	18.7
0	77.9	31.9	56.6	31.6
1	9.4	13.0	18.6	13.7
2	1.0	7.1	2.9	7.2
3	0.2	3.5	0.6	4.3
4	0.2	2.6	0.4	2.4
5	0.0	1.5	0.3	1.0
6	0.0	1.1	0.1	1.0
7	0.0	0.5	0.0	0.6
8	0.0	0.1	0.0	0.1

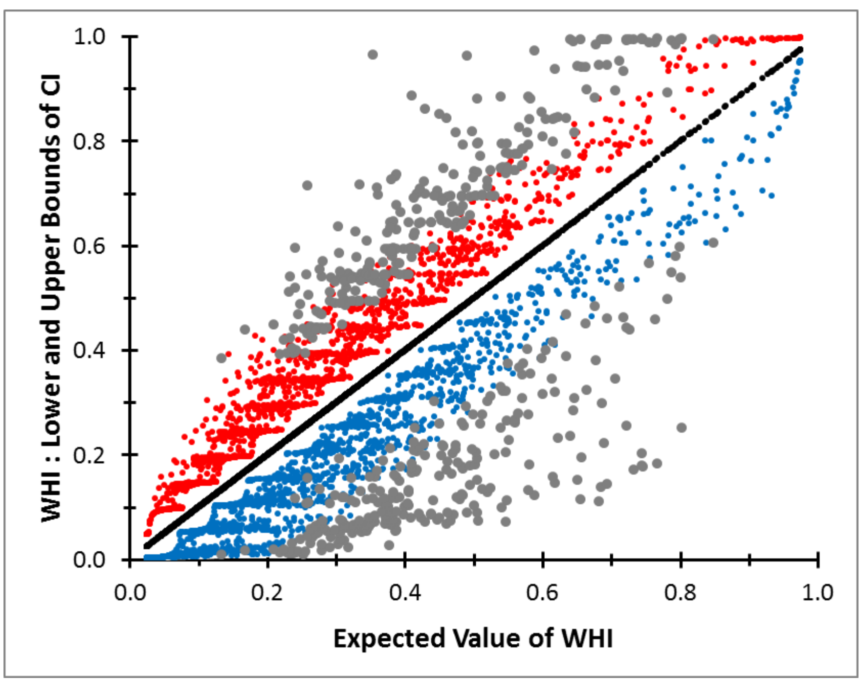


Figure 3.13. Uncertainty of WHI values. Black points are expected value of WHI. Red and blue points are lower and upper bounds of 90% confidence interval, respectively. That is, the bounds contain 90% of WHI values generated in the uncertainty analysis. Gray points represent 10% of AUs with widest confidence intervals.

3.4 Discussion

We developed an index that quantifies the relative value of places for the conservation of native freshwater animal species. The main application of this assessment is land use planning, and land use plans should use the results to direct residential development to places that will minimally impact lotic habitats. The first places to develop or develop more densely are those AUs with the lowest scores for relative conservation value (Figure 3.10). Rural development should be avoided in AUs at the highest end of relative conservation value and occur first in AUs that have the lowest relative conservation value.

Our assessment shows that relative conservation value or place-based conservation priorities cannot be conveyed by one map. The relative value of a place depends on one's perspective, e.g., preferences for "quality" or "quantity," and therefore, it is wise to examine maps for multiple perspectives. Also, the appropriate management of a place (i.e., AU) may depend on the roles it serves for the conservation of lotic habitats: wetlands and floodplains, local salmonid habitats, effects on downstream habitats, or some combination. Seventy percent of AUs had a moderate to high value for at least one component of the index, and therefore, warrant management that will at least maintain that level of relative conservation value.

Our assessment highlights the challenges faced by county governments trying to conserve lotic habitats and salmonid habitats, in particular. The assessment indicates that a large proportion of almost every WRIA has moderate to high relative value for some component related to the conservation of lotic habitats. Furthermore, our assessment shows that rural areas outside public lands have the highest density of extant wetlands and undeveloped floodplains in the Puget Sound Basin (Figure 3.14). The main reasons for this are that most wetlands in urban areas have been filled and public lands are mostly confined to montane areas which have naturally low wetland density. This finding is not surprising and probably well known by most county land use planners. Our assessment also shows that a band of rural lands between urban areas and the Cascades or Olympic mountains still have at least moderate ecological integrity (Figure 3.15). Again, this finding is not surprising. Maintaining and enhancing aquatic ecological integrity in these rural areas may pose challenges for local governments. These two results, based on two simple analyses, emphasize the value of private rural lands for the conservation of lotic habitats.

Our conceptual model emphasizes longitudinal and lateral connectivity. Upstream longitudinal connectivity was incorporated into our aquatic ecological integrity index. Downstream longitudinal connectivity was assessed with the downstream accumulative habitats index. For landscapes dominated by public lands (≥ 80% public lands), over one-third (36%) of the land area had a high score (≥0.8) for downstream accumulative habitats and over four-fifths of the land area had at least a moderate score (≥0.5). In other words, as is already well-known, public lands in the Puget Sound Basin affect substantial amounts of salmonid habitats through the headwaters flowing from them. The correlation between the percent of public land in an AU and its downstream accumulative habitats score was 0.59. In contrast, the correlation between the percent of public land in an AU and its hydrogeomorphic features score was -0.14. For AUs dominated by private lands (≤ 20% public lands), nearly one-quarter (23%) had a high score (≥0.8) for hydrogeomorphic features and half had at least a moderate score (≥0.5). These facts indicate that in many landscapes dominated by private lands lateral connectivity should be of primary concern

County governments have regulatory authority over land use on private lands in rural areas. The allowed land uses designated through local government comprehensive plans and shoreline master programs may be the most important actions affecting the health of lotic habitats. Maintaining wetlands, floodplains, and ecological integrity in rural areas while accommodating human population growth will require sophisticated approaches to land use planning and residential development.

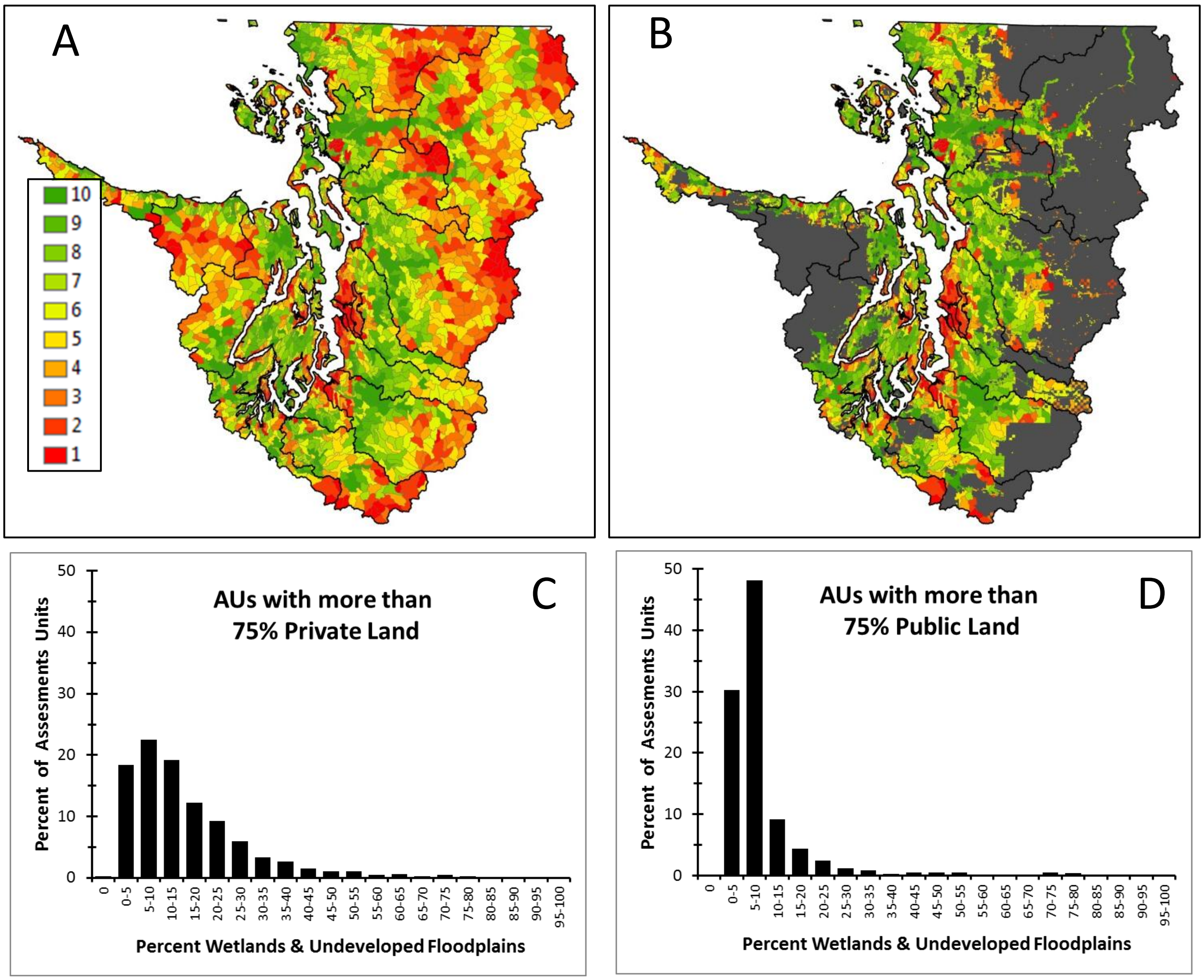


Figure 3.14. Hydrogeomorphic features outside public lands. (A) The density of extant wetlands and undeveloped floodplains. (B) Map A overlaid with public lands (dark gray). (C)The distribution of hydrogeomorphic feature density for AUs with more than 75% private lands, and (D)for AUs that are more than 75% public land. The distribution for private lands has a greater proportion of AUs with hydrogeomorphic feature density above 20%. When public lands are overlayed on the Map A most of the red and orange AUs are covered and mostly green AUs remain.

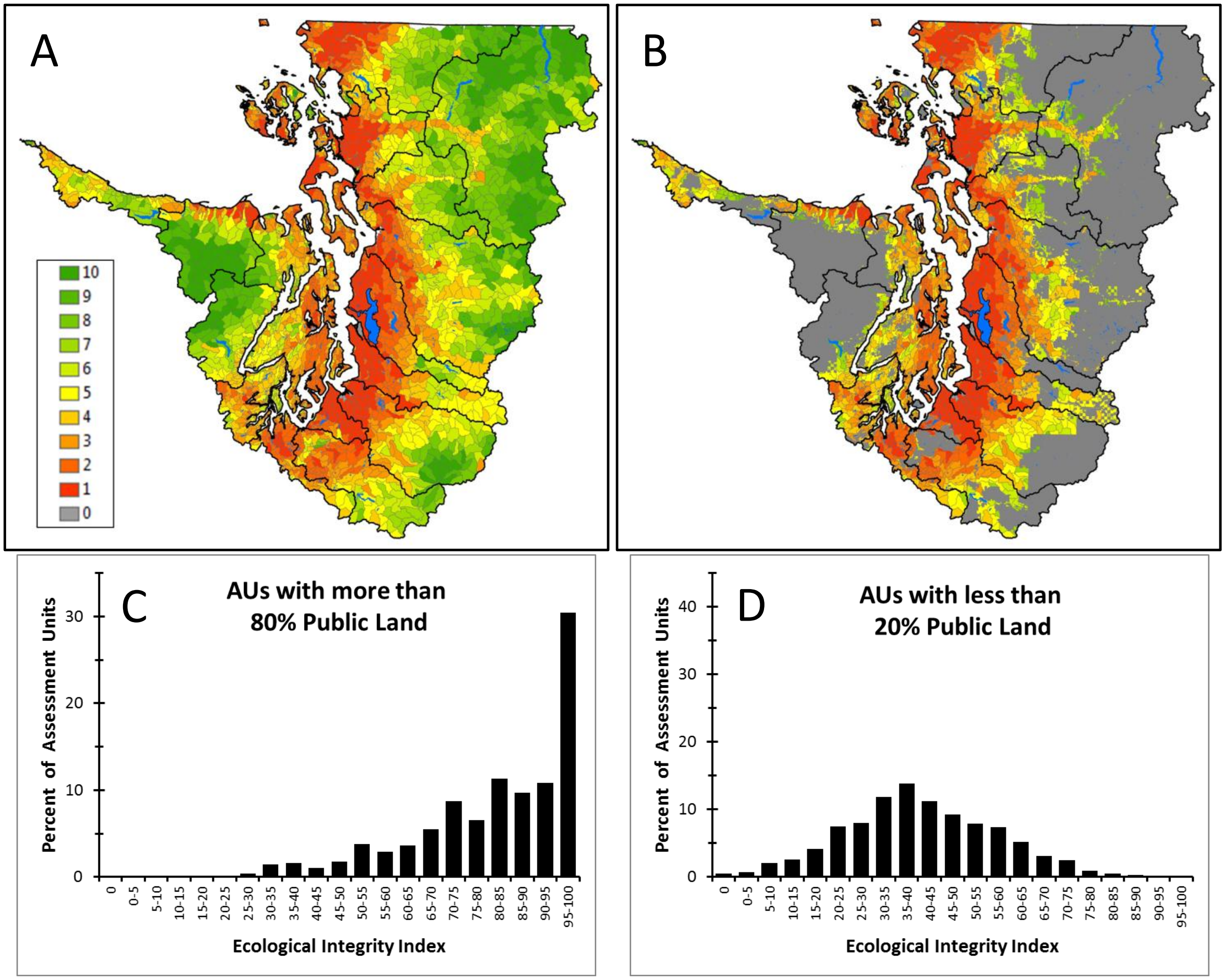


Figure 3.15. Relative ecological Integrity outside public lands. Map A shows relative ecological integrity for the entire Puget Sound Basin. Map B is Map A overlaid with public lands (dark gray). Graph C is for AUs that are more than 80% public land. Graph D is the distribution of ecological integrity values for all AUs with less than 20% public land. Almost no AUs dominated by private land have high ecological integrity (>0.80). However, many AUs dominated with by private lands have moderate integrity (between 0.3 and 0.7) that should be maintained or enhanced.

3.4.1. Caveats

The spatial extent of our assessment covers the entire Puget Sound Basin, however, we did separate assessments for each of 18 WRIAs in the Basin. *Index scores cannot be compared across WRIAs*. Our assessment in its current form does not enable Basin-wide comparisons. Regional authorities should keep that in mind when using this assessment. The assessment may be useful for establishing regional priorities for the protection of rivers and streams, but be aware that comparisons are only valid within WRIAs.

Our indices indicate the quality, amount, and richness of habitats for those particular species and species' life history variants for which we collect data. In freshwater environments, government agencies generally collect occurrence data for harvested species, and, in particular, the harvested life stages of those species. In other words, we have reasonably accurate occurrence data for adult salmonids but we lack accurate occurrence data for the juvenile life stages of salmonids and for the majority of other animal species that inhabit rivers and streams. Nevertheless, as explained above, we believe that salmonids can effectively serve as umbrella species for all other species that rely on lotic habitat types. The umbrella species approach has one obvious shortcoming – some species may be limited by ecological factors not relevant to the umbrella species (Roberge and Angelstam 2004). Hence, our indices may not be a comprehensive assessment of relative conservation value for all animal species found in the rivers and streams of Puget Sound.

As stated above, our data on salmonid species presence are based mostly on adult presence. Other life stages are poorly represented in the data. Many coastal inlets are associated with small streams that do not contain adult salmon species but may support juveniles (E. Beamer, Skagit River System Cooperative, pers. com.). Thirty-four percent of our AUs are "coastal" AUs located on the shoreline of Puget Sound. Of these small AUs (mean size 1 square mile), 22% are unlikely to contain a stream, 25% contain a stream known to be inhabited by at least one salmonid species, and 51% contain a stream that, according to our data, is not inhabited by salmonids. Some AUs in the third group of coastal AUs may, in fact, support juvenile salmonids.

3.4.2. Potential Improvements

Our indices could be improved several ways. First and foremost, more up-to-date and accurate fish occurrence data are needed, including occurrence data on juvenile salmonids. At present, the Northwest Indian Fisheries Commission and WDFW maintain separate databases for salmonid freshwater occurrences. The formats and structures of the two databases are incompatible, however, the two agencies are currently working together to merge their databases. We used the WDFW database for this assessment. The merged database should be more comprehensive and accurate than the separate databases. After the database merger is completed our freshwater assessment could be redone.

The umbrella species approach was motivated, in part, by the lack of accurate occurrence data for other freshwater species. WDFW should expand data collection beyond harvested fish species. More occurrence data would also lead to increased understandings of the habitat associations of lamprey, sculpin, dace, and sucker species that could be incorporated into future conservation assessments.

Second, we need models of salmonid habitat quality that can be applied regionally. Salmon will always be a major driver of freshwater conservation in the Puget Sound Basin. Therefore, salmonid habitats must be components of any freshwater habitat conservation assessment. We constructed salmonid habitat models by combining IP models with an index of aquatic ecological integrity. IP models and ecological integrity indices are based largely on expert opinion. We relied on expert opinion because we lacked empirically-based statistical models for salmonid habitat quality. Currently available models are watershed-specific and exist for only a few WRIAs in Puget Sound Basin. The use of expert judgment as a substitute for empirical information has been criticized (Ruckelshaus et al. 2002), but we had no other practical alternative. Empirically-based statistical models that can be generalized to the entire Puget Sound Basin or to substantial portions of the Basin should be a major focus of future research.

Third, we developed our own index of aquatic ecological integrity but there are many other formulations of aquatic ecological integrity which may be better (e.g., Quigley et al. 2001, Mattson and Angermeier 2007, Theobald et al. 2010). Validation of our ecological integrity index could be done be comparing it to these other independently derived indices. All of these indices, including ours, were based mainly on expert opinion, consequently, high correlation among indices would, in effect, constitute agreement amongst independent groups of experts. The aquatic ecological integrity index might also be improved by developing an empirically-based statistical model that relates the composition of aquatic communities (e.g., some combination of the metrics that comprise currently available benthic macro-invertebrate and fish IBIs) to various metrics of landscape-scale watershed condition

Further discussion of the assessments is provided in Part 5 of this report.

Part 4:
Marine Shoreline Habitats Assessment

4.1. Conceptual Model

A conceptual model is a simplified representation of a complex system that emphasizes the interrelationship of the major elements rather than the details of each element. The conceptual model forms the basis for the components and structure of the quantitative model.

4.1.1. Conceptual Foundation

We begin with a conceptual model of ecosystems in which processes and structures[14] interact to manifest functions (Figure 4.1; Goetz et al. 2004, Simenstad et al. 2006b). Maintaining both process and structure is essential to the maintaining healthy nearshore ecosystems. The composition and organization of biological communities in nearshore ecosystems is caused by processes such as wave exposure, sediment suspension, and freshwater flows, and by "structures" such as beach topography, beach sediments, and salinity. The structure of nearshore ecosystems is both the consequence of and an influence on the action of ecosystem processes (Goetz et al. 2004). For instance, beach topography is the result of wave action but beach topography also influences wave action.

In this assessment we focus on a specific set of nearshore ecosystem functions: the habitats for nearshore flora and fauna (Figure 4.2). The habitat functions of nearshore ecosystems are highly integrated and hierarchical. For example, the ecosystem function of herring[15] habitat depends in part on the presence of a particular vegetative structure, eelgrass (*Zostera marina*), and the ecosystem function of eelgrass habitat depends largely on the structure provided by beach sediments (Figure 4.3).

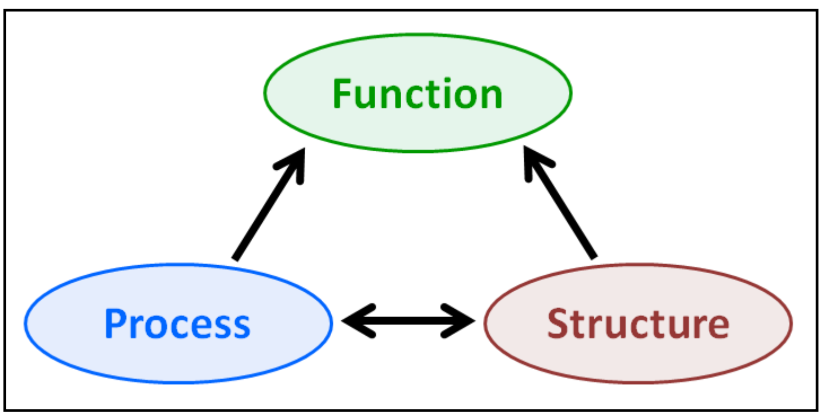


Figure 4.1. Ecosystem processes and structures interact to manifest ecosystem functions such as the provision of habitat (Goetz et al. 2004).

Habitat Shaping Processes and Structures

The dominant physical process along the shorelines of Puget Sound is the movement of sediments. Sediment movement occurs within spatially distinct littoral drift cells. Drift cells are comprised of

[14] Other ecosystem conceptual models separate structure into structure and composition. We consider composition to be an attribute of structure.

[15] Scientific names of animals listed in Appendix E.

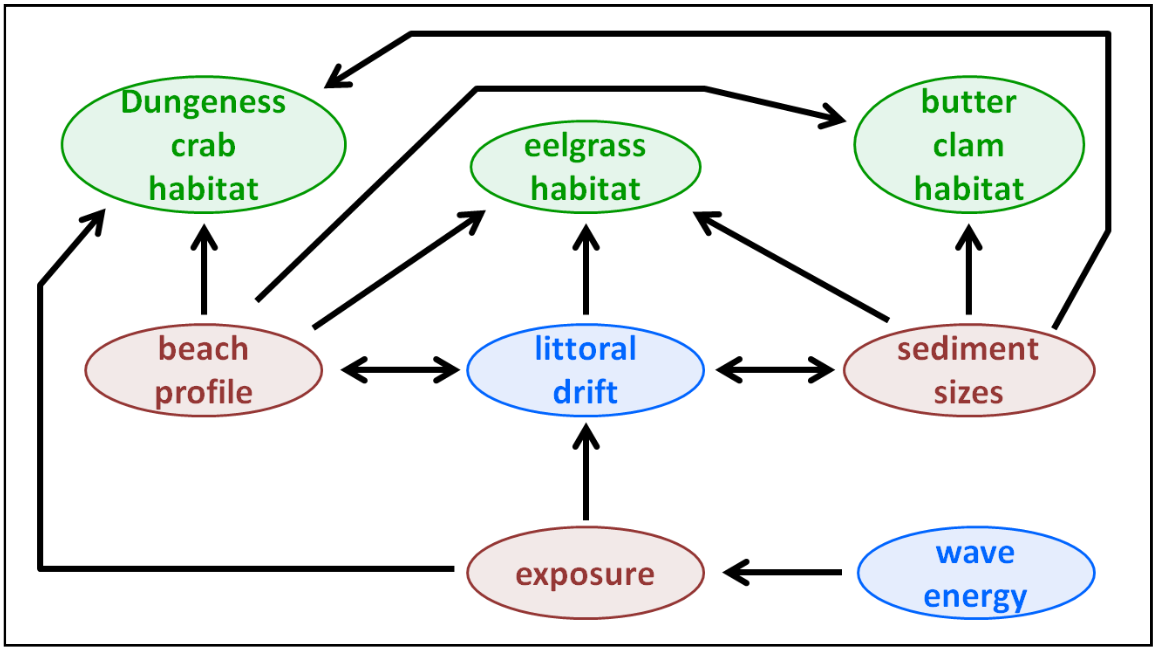


Figure 4.2. Interaction of processes (blue ovals) and structures (brown ovals) produce habitat functions (green ovals). Each species has different habitat requirements that are met by different interactions among processes and structures.

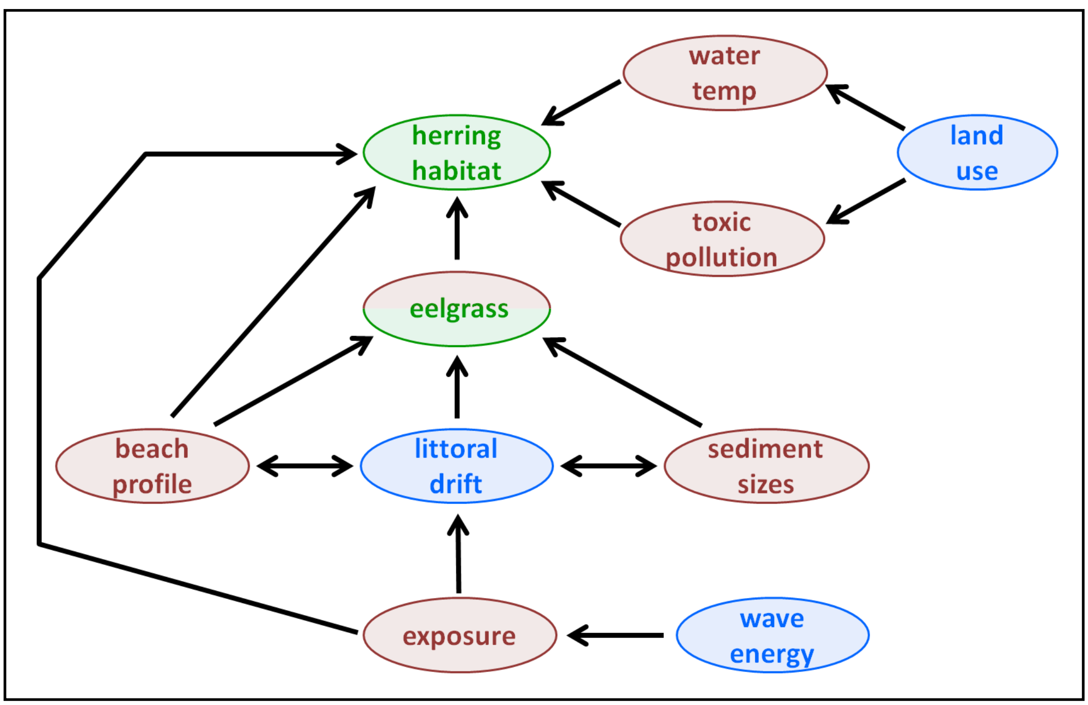


Figure 4.3. Hierarchical nature of ecosystem structures. Habitats for herring and eelgrass depend on the interaction of processes (blue oval) and structures (brown ovals). Eelgrass depends on the structure provided by sediments and beach profile (i.e. topography). Herring depend on the structure provided by eelgrass. Hence, eelgrass represents both an ecosystem function (i.e., a habitat function) and a structure.

sediment sources, typically bluffs, where erosion provides sediment for beaches, sediment sinks, where sand and gravel accumulate, and transport reaches where littoral drift connects sources to sinks (Figure 4.4). Puget Sound's shorelines are comprised of 812 drifts cells with an average length of 3.7 miles and a maximum of length 60 miles.

Within drift cells, variation in wave exposure, sediment sources, and local geomorphology create a variety of shoreforms (Table D5), such as bluff-backed beach, barrier estuary, open coastal inlet, and rocky shore (Shipman 2008). Many bluff-backed beaches are sediment sources, barrier beaches and barrier estuaries are sediment sinks, and all beaches play a role in sediment transport. Shoreforms provide a variety of environmental settings for fish and wildlife habitats. At a finer spatial scale, ecosystem structures have been mapped as shorezones (Berry et al. 2001b). Shorezones classify shorelines according to sediment type, slope, and wave exposure (*sensu* Dethier 1990 and Howes et al. 1994).

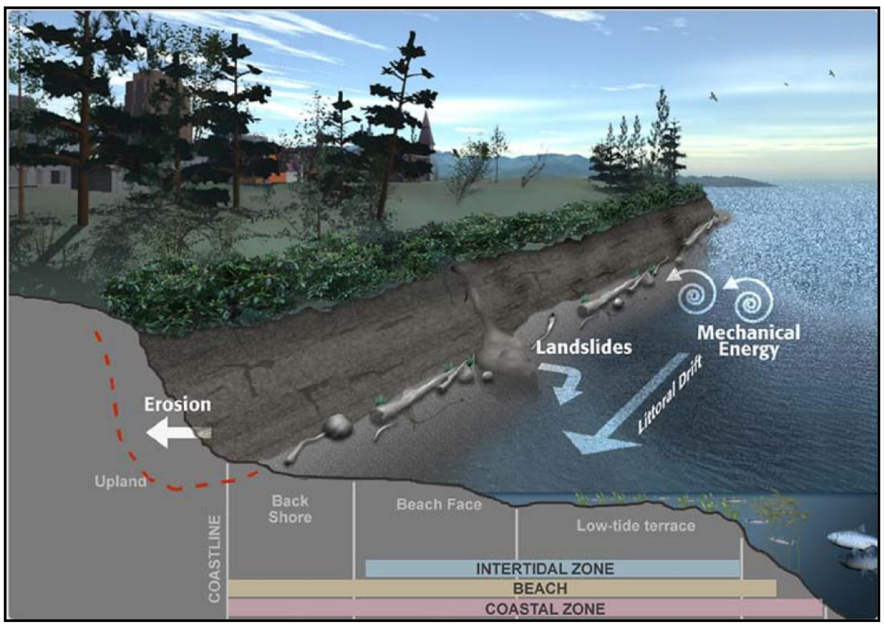


Figure 4.4. The physical process of sediment movement within littoral drift cells along the shorelines of Puget Sound (from Simenstad et al. 2006a)

Littoral drift is the dominant process for shaping and maintaining shoreline habitats. Therefore, in order to fully assess the quality of shoreline habitats, the integrity of drift cells must also be assessed. *Ecological integrity* has been defined as the ability of an ecological system to support and maintain a community of organisms that has species composition, diversity, and functional organization comparable to those of natural habitats within a region (Parrish et al. 2003). An ecological system has integrity when its dominant ecological characteristics (i.e., structures, processes, and functions) occur within their natural ranges of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human disruptions.

Littoral drift is not the only ecosystem process affecting habitat functions along the shorelines of Puget Sound. For instance, within the nearshore zone species may be directly affected by wave exposure, and just outside the nearshore, species are affected by upwelling currents. Structures in close proximity to the nearshore zone may also influence habitat quality along shorelines. For instance, rocky reefs in subtidal areas affect foraging habitat quality for white-winged scoter and patches of large trees in upland areas affect nesting habitat quality for great blue herons. Assessing the condition or presence of the myriad structures and processes that affect habitats in the nearshore zone was beyond our capability. Therefore, we assessed the quantity of habitat functions as indicated by the presence of species.

Habitat Functions

A vital ecosystem function is the provision of habitats. Habitats are specific to each species. Although there is considerable overlap in the habitat characteristics of some species, e.g., red sea urchins and green sea urchins, the full multi-dimensional characteristics of habitat are unique for every species and sometimes unique to a particular population of a species (Morrison et al. 1992). The behavioral and evolutionary processes that manifest a species' habitats are highly complex: individuals integrate multiple factors when selecting habitat, exhibit a wide range of habitat preferences, respond differently to habitats with different qualities, and populations are adaptable to changing habitat conditions. Hence, for many species our understanding of habitat is simplistic, and consequently, our ability to model species' habitats, habitat quality, or habitat functions is limited and replete with uncertainty.

Mapping species-specific habitats is technically challenging, but mapping the presence of a species is not (although is it fiscally challenging), and considerable expense has been invested in mapping the presence of certain marine species, in particular, harvested and imperiled species. By definition, the presence of a species establishes the presence of that species' habitat. In other words, if a species is present at a site, then that site is serving a habitat function. However, habitat quality cannot be determined by species presence alone, and the functions (e.g., breeding, rearing, resting) served by that habitat may not be discernible through species presence. Furthermore, species absence does not establish the absence of habitat; species absence may be due to survey error, patterns of seasonal use, or declining population size. Nevertheless, considering the dearth of species-specific habitat models, the presence of a species is our most reliable indicator of habitat.

Collection of empirical data by WDFW and other agencies on the locations of fish or wildlife species generally focuses on imperiled or harvested species. For the vast majority of other species, site-scale location data are based on incidental observations or incomplete surveys. These data have a high rate of omission error, i.e., false negatives. For many vertebrate species comprehensive data on locations are available as range maps (e.g., Wahl et al. 2005), but these can be highly inaccurate at spatial scales of about 4 square miles or more.

4.1.2. A Model for Relative Conservation Value

Of the three habitat assessments conducted for the Puget Sound Partnership's Watershed Characterization Project – terrestrial, freshwater, and marine shorelines − the fish, wildlife, and habitat data for marine shorelines are the most comprehensive and very likely the most accurate. This can be attributed to the one-dimensional nature of shorelines and their relatively small spatial extent − 2,468 miles of marine shoreline in Puget Sound compared to over 50,000 miles of rivers and streams in Puget Sound Basin.

Given the quality of the data, we believed an assessment based on the presence of species would provide a credible indicator of conservation value. The overarching assumption of that belief is that the relative value of shorelines for the conservation of fish and wildlife habitats is predominantly a function of the presence of the species and habitats for which we collect occurrence data. In general, we collect occurrence data for certain species or habitat types because 1) humans harvest those species, 2) we are concerned about the status of those species or habitats (e.g. threatened or endangered species), or 3) we are concerned about the management of those species or habitats (e.g., species highly sensitive to human disturbances). In other words, we collect data on those species and habitats we care most about. Therefore, an assessment based on these data should indicate those places we should care most about for the conservation of fish and wildlife habitats.

Another major assumption is that the relative value of a place for the conservation of fish and wildlife habitats can be accurately quantified. And, more specifically, that relative value can be expressed through a single comprehensive number – an index (or a small set of indices). Furthermore, we assumed that the index is a linear function – the weighted linear combination of normalized biological data. Better relationships between relative conservation value and the biological data may exist, but lacking any practical means to determine those relationships we chose the most parsimonious formulation – a linear equation.

4.2. Methods

4.2.1. Spatial Framework

Puget Sound has been divided into 7 oceanographic sub-basins based on bathymetry and circulation patterns, however, these sub-basins also reflect regional patterns in shoreline geology, geomorphology, and wave environment (Shipman 2008). For instance, the South Puget Sub-basin contains no rocky shoreforms (rocky platform/ramp or plunging rocky) but 47% of shorelines in the San Juan Sub-basin are rocky. Over half of all closed lagoon marsh shoreforms are in the North Central Sub-basin but the South Central, Hood Canal, Whidbey, and South Puget sub-basins have no closed lagoon marshes. Also, 37% of shorelines in the Whidbey Sub-basin are located in river deltas but all other sub-basins have 6% or less of their shorelines in river deltas. These regional differences in shoreforms manifest regional differences in biological communities. Given these and other regional differences in shoreforms, we minimized comparisons of dissimilar communities by doing the assessment within sub-basins.

Puget Sound has 2,468 miles of marine shoreline. For purposes of analysis, this shoreline must be broken into smaller spatial units. We intersected the shoreform (Shipman 2008) and shorezone (Berry et al. 2001a, 2001b) classification systems to produce a shoreline composed of 10,178 segments (Table D6). Demarcations between segments correspond to observed changes in shoreform (e.g., barrier estuary, bluff-backed beach), morphology (e.g. flat, platform, ramp), or substrate (rock, gravel, sand), all of which significantly influence plant, fish, and wildlife habitats. Across the entire Puget Sound, the mean segment length was 0.24 miles and 75% of segments were less than 0.29 miles (Table 4.1). Mean segment lengths were different among the seven oceanographic sub-basins, with the Juan de Fuca Sub-basin having the longest (0.55 miles) lengths and the San Juan Sub-basin having the shortest (0.17 miles).

In the geographic database a shoreline is one-dimensional. Shoreline segments were converted to two-dimensional polygons for two reasons. First, for some species, such as Dungeness crab, the spatial extent of each occurrence was represented as a polygon and we wanted to maintain that two-dimensional information. Second, there were fish and wildlife occurrence data, such as those for bald eagle nests, that did not intersect the shoreline but were in close proximity to it and we wanted to associate those data with the shoreline. To accomplish both objectives we buffered the shoreline by approximately ¼ mile[16] in the landward direction and by approximately ¼ mile or extreme low tide, whichever was farther, in the seaward direction (Figure 4.5). The National Wetlands Inventory (USFWS 1989, Cowardin et al. 1979) inter-tidal polygons were used to delineate the location of extreme low tide. Biological data outside the ¼ mile buffer were excluded from the analysis.

One-quarter mile was chosen as the buffer width because we believed it would encompass most fish and wildlife resources that might be impacted by shoreline development. One-quarter mile[17] is approximately the recommended management zone around bald eagle nests and roosts (Watson and Roderick 2000) and ¼ mile is roughly the distance at which shoreline development might disturb seal and sea-lion haul outs (S. Jeffries, WDFW, pers. commun.). Furthermore, for at least 90% of the Puget Sound shoreline, ¼ mile encompasses the entire nearshore zone (< 10 m depth, *sensu* Simenstad et al. 2011) and most shallow subtidal areas.

---

[16] The actual distance was exactly 400 m. We describe it as approximately ¼ mile in order to consistently use English units of measure throughout this report. ¼ mile equals 402 m.

Table 4.1. Summary of shoreline segment sizes by oceanographic sub-basin. Units in miles.

sub-basin	N	Mean	min	1st qtr	median	3rd qtr	max
Juan de Fuca	372	0.55	0.029	0.12	0.29	0.67	6.75
San Juan	4353	0.17	0.003	0.04	0.09	0.19	8.24
Hood canal	874	0.28	0.004	0.09	0.17	0.34	3.61
Whidbey	1014	0.34	0.004	0.10	0.21	0.38	7.36
North Central	393	0.32	0.010	0.12	0.24	0.42	2.44
South Central	1482	0.25	0.004	0.09	0.18	0.31	1.99
South Sound	1690	0.26	0.005	0.09	0.17	0.34	3.86
ALL	10178	0.24	0.003	0.07	0.14	0.29	8.24

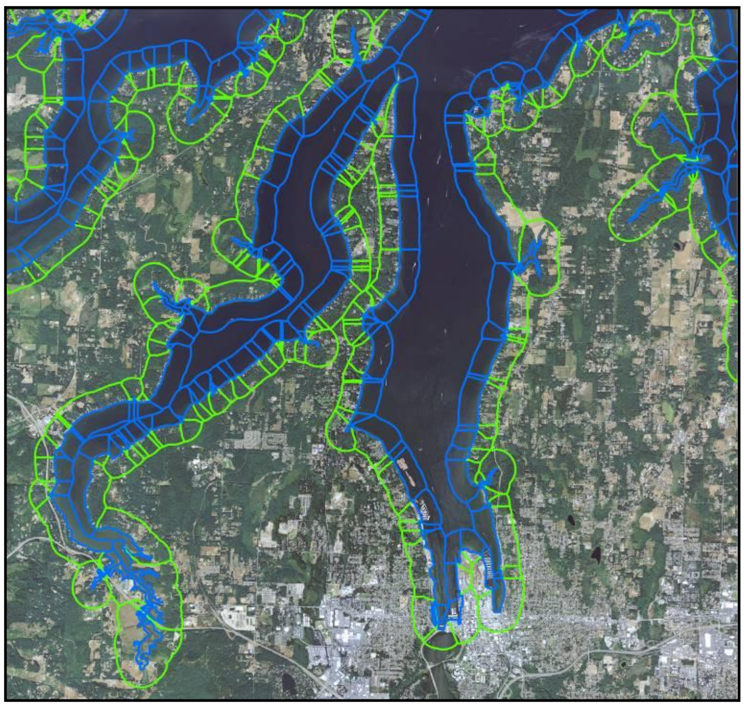


Figure 4.5. Spatial framework for the shoreline habitats assessment. Blue and green polygons are seaward and landward assessment units, respectively. This example depicts Totten, Eld, and Budd Inlets (left to right) in the South Sound Oceanographic Sub-basin.

4.2.2. Data and Data Processing

The Biological Data

We are limited by breadth, precision, and accuracy of the biological data. The data's breadth, i.e., the variety of species and habitats for which occurrence data are available, is relatively broad: eight shellfish[17] species or species groups of commercial/recreational interest, urchins, three forage fish species, eight salmonid species, numerous bird species, pinnipeds, kelp, eelgrass, surfgrass, and wetlands (Figure 4.6). The measurement precision for most of these data is at the level of presence/absence. Only PSAMP bird survey data enable an estimate of local density. The accuracy of our data is affected by the data's age and the methods of data collection. Some data sets are over 20 years old (e.g., WDF 1992). Most data were collected through field surveys, but some data in certain datasets are "based on 'best professional judgment' of the biologist."

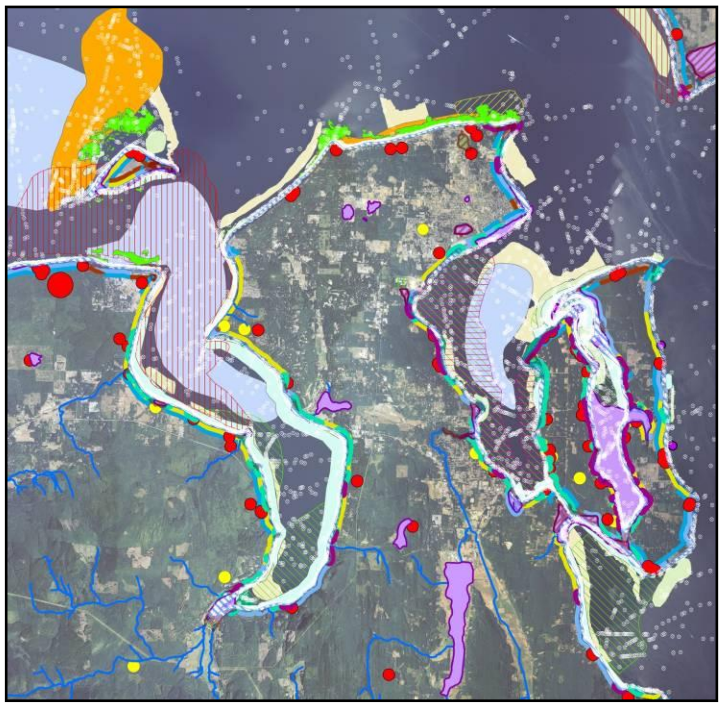


Figure 4.6. Biological data used in shoreline habitats assessment. Different polygon colors and fill patterns and different line colors represent different plant, fish, or wildlife species. White dots are observations of PSAMP bird surveys. This example depicts Discovery Bay and Port Townsend Bay.

We reviewed all biological datasets managed by WDFW for their relevance to marine shorelines in Puget Sound and their likely accuracy. Our subjective evaluation of likely accuracy considered the dataset's age, how the data were collected, and the detectability of the taxa surveyed. Occurrence data for fish and wildlife are more prone to false negatives than to false positives, and hence, we were particularly concerned about the potential frequency of false negatives in each dataset. We settled on 41 data sets (Tables 4.2 and 4.3). Most data sets mapped the occurrences of single species (i.e., Dungeness crab, herring). Some data sets mapped the simultaneous occurrence of multiple species (i.e., shorebird and

[17] Shellfish includes both mollusks such as butter clam, and crustaceans such as Dungeness crab.

waterfowl concentrations). For some data sets which were likely to have a high rate of false negatives we developed models for relative likelihood of occurrence (explained below). Table D1 lists known data sets that were excluded from the assessment.

With a few exceptions the fish and wildlife species included in the assessment were priority species as designated by WDFW's Priority Species and Habitats program (PHS; WDFW 2008). Priority species require protective measures for their survival due to their population status, sensitivity to habitat alteration, and/or recreational, commercial, or cultural importance. Priority species include State Endangered, Threatened, Sensitive, and Candidate species; and animal aggregations considered vulnerable (e.g., heron colonies, shorebird concentrations).

We also included a subset of the "bio-band" data from DNR's shorezone database (Berry et al. 2001b). These data are referred to as bio-bands because certain plants and animals create a well-defined series of cross-shore color bands. Each bio-band is named for the most prominent species in the band or by the general description of the species assemblage. The abundance of each band is recorded as either absent, patchy, or continuous, which we translated to 0, 1, or 2. We included bio-bands for species of concern (e.g., eelgrass, kelp) and excluded common species (e.g., barnacles, sand dollars).

The highest quality data utilized in our assessment was that collected by the Puget Sound Ambient Monitoring Program (PSAMP; Nysewander et al. 2005). PSAMP has conducted highly systematic aerial surveys of birds on Puget Sound since 1992. The complete data set contained 381,214 observations; over half the observations are of multiple birds. We removed records that were older than 1995, summer surveys, non-marine birds (e.g., common raven, northern flicker), or extremely abundant birds (e.g., glaucous-winged gull). This filtering process reduced the data set to 196,312 observations and included 65 bird species (Table D2) and observations for 31 categories of partially identified birds (e.g., "Unidentified Diving Duck"; Table D3). We summarized the data by calculating two indices. First, for the years 2000 to 2009, we calculated the median density of birds for each shoreline polygon. Second, for "at-risk" species (Table D4) we calculated average density per shoreline polygon over the years 2005 to 2009.

Relative Likelihood of Occurrence

We believed the spatial data for some species likely had a high rate of false negatives. We were particularly concerned about data collected by the Washington Department of Fisheries in the 1980s (WDF 1992). To compensate for the shortcomings of these data we developed relative likelihood of occurrence (LO) models. The shoreform-shorezone classification system was treated as habitat types and we calculated $L(S|H)$, the relative likelihood that a species is present given the presence of a particular habitat type.

We had hoped to develop probability of occurrence models which would provide more robust estimates of species occurrences, however, the data precluded such models because: 1) the species data were not collected through a random sampling of shorelines, and 2) the data do not record negative surveys, and hence, there is no way to distinguish between species absence at a location and a lack of survey effort at that location. Each relative likelihood model is effectively a "presence/absence index" based on a species' degree of association with each habitat type.

Table 4.2. Summary of fish and wildlife data used in the indices.

Taxon	PHS[1]	Occur. model	Description	Units	Source
Northern abalone	X	X	documented occurrences	Square feet	WDF 1992
Clams; intertidal hardshell	X	X	beds that could be commercially harvested or have significant recreational usage		WDF 1992
Clams; subtidal	X	X			
Dungeness Crab	X	X			
Pacific oyster	*X*	X	non-native *Crassostrea gigas*		WDF 1992
Geoduck	X	X	beds that could be commercially harvested		WDF 1992
Pandalid shrimp	X	X	pink, coonstripe, and spot shrimp		WDF 1992
Sea Urchins	X	X	documented occurrences of red and green sea urchins		WDF 1992
Herring Holding Areas	X		where adults congregate each winter prior to spawning	Square feet	WDFW 1992
Herring Spawning Areas	X		regular surveys over 10 years		WDFW 2000-2009
Surf smelt	X	X	data represent more than 30 years of spawning beach surveys	feet	WDFW 1972-2008
Pacific Sand lance	X	X			WDFW 1972-2008
Bull Trout	X		number of stream mouths inhabited by species that intersect shoreline segment	count	WDFW Fishdist
Chinook Salmon	X				
Chum Salmon	X				
Coastal Cutthroat	X				
Coho Salmon	X				
Pink Salmon	X				
Sockeye	X				
Steelhead Trout	X				
Bald Eagle Communal roosts	X		zone around roost site; radius = 400 m	Square feet	WDFW WSDM[2]
Bald Eagle nest	X		zone around nest site; radius = 200 m		
Great Blue Heron colonies	X		zone around occurrence point; radius = 1000 ft		
Black Oystercatcher nests			survey data from 2010	count	
Shorebird	X		large regular concentrations	Square feet	
Waterfowl	X		large regular concentrations		
Important Bird Areas (IBA)			support species of concern or high densities of birds		Audubon 2001
Bird Density			median density of all birds from 2000 to 2009 (Tables D3 & D4)	birds / km^2	WDFW PSAMP[3]
"At-Risk" Bird density			density of "at risk" birds from 2005 to 2009 (Table D5)		
Seal/sea lion haul-out	X		both natural (e.g., islands) and artificial (e.g., buoys) haul outs for harbor seals and California sea lions	count within 400 m of shore	WDFW WSDM

[1] PHS: Is on the WDFW's priority habitat and species list
[2] WDSM: WDFW's Wildlife Survey Data Management
[3] PSAMP: Puget Sound Ambient Monitoring Program

Table 4.3. Summary of plant and wetland data used in the indices.

Taxon	Description	Units	Source
dune grass	salt-tolerant grasses, dominated by *Leymus mollis*	Amount = shoreline length *x* bioband density Density 0 = Absent; 1 = 0-50% cover; 2 = 50-100% cover	DNR Shorezone (Berry et al. 2001a, 2001b)
sedges	brackish/ freshwater wetlands assemblages; found at freshwater streams and river mouths		
high salt marsh	brackish/ freshwater wetlands assemblages; *Triglochin*/*Salicornia*/ *Deschampsia*/*Distichlis*		
low salt marsh	dominated by *Salicornia*		
surfgrass	*Phyllospadix* spp. of lower intertidal		
eelgrass	*Zostera marina* and introduced *Z. japonica*		
brown kelp	large bladed *Laminaria* / *Saccharina* spp.		
chocolate brown kelp	*Laminaria setchellii*, *Eisenia* and/or *Pterygophora*, *Hedophyllum*, *Egregia*		
bull kelp	*Nereocystis luetkeana*		
giant kelp	*Macrocystis spp.*		
wetlands (NWI)	all wetlands except marine sub-tidal	square feet	USFWS 1989

The shoreform-shorezone data are a complete census of the Puget Sound shoreline. Therefore, the probability of a particular shoreform-shorezone habitat type, H, occurring at a randomly selected shoreline segment is:

$$P(H) = n_H / M \tag{4.1}$$

where M is the total number of shoreline segments in Puget Sound and n_H is the number of shoreline segments in Puget Sound classified as habitat type H. The relative likelihood that a species is present given the presence of a particular habitat type is:

L(S | H) = (relative frequency of species S in habitat H) / (relative frequency of species S in Puget Sound)

which is calculated as:

$$L(S \mid H) = (n_{SH}/n_H) / (n_S/M) \tag{4.2}$$

where n_S is the number of shoreline segments in Puget Sound where species S is recorded as present and n_{SH} is the number of shoreline segments that were classified as habitat type H and species S is recorded as present.

We developed LO models for 10 species (Table 4.2). The relative likelihood of occurrence was based on only one variable – the habitat types created through the intersection of shoreforms (Shipman 2008) and the habitat types (Dethier 1990) in the DNR shorezone data (Berry et al. 2001b). Our simple LO models are unlikely to make highly accurate predictions of species occurrences. In fact, we believed that the LO models would be highly biased toward false positives, i.e., predicting a habitat association where there is none. However, we also believed that a simple model with a high rate of false positives was preferable to inaccurate data with a high rate of false negatives. Using the model was a precautious approach.

The LO model results were merged with the occurrence data; empty records in the occurrence data were substituted with the relative likelihood of occurrence and data records with known presence were set to a likelihood of 1.

Data Normalization

Our species data come in numerous forms: linear units, areal units, counts, density, presence/absence, and categorical (Tables 4.2 and 4.3). To avoid unintended weighting of one variable more than another when combining these data we must convert them to commensurate units. The first conversion was to density – data in linear or areal units were divided by the length or area, respectively, of their corresponding shoreline segment. Salmon and seal/sea lion haul-out occurrence data were not converted to density; they remained as counts. Data for species that had LO models remained in likelihood units. The second conversion was a normalization. In effect, we converted data which were originally in nominal, interval, and ratio scales to a common form of ratio scale. These densities, counts, or likelihoods were normalized from 0 to 1 within oceanographic sub-basins using the following equation:

$$N(v_{Sj}) = (v_{Sj} - V_{min}) / (V_{max} - V_{min}) \qquad (4.3)$$

where v_{sj} is the value for species or species group *S* at shoreline segment *j*, V_{min} is the smallest value for species S in the sub-basin, and V_{max} is the largest value for species *S* in a sub-basin.

4.2.3. The Index

Indices of Conservation Value

We want to quantify the relative habitat value of marine shorelines. There are myriad formulations for a quantitative index, each with their own particular advantages and disadvantages. We limited our assessment to two simple formulations based on two perspectives of relative conservation value that reflect the quantity versus quality dichotomy. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's most significant contribution, i.e., the quality a place contributes.

The first perspective can be implemented by summing the amount of habitats at each shoreline segment. That is, summing the normalized counts or densities of each species or species group for each shoreline segment. The *composite index* of relative habitat value for a shoreline segment *j* is:

$$AV_j = \sum_{S=1}^{T} w_S N(v_{Sj}) \qquad (4.4)$$

where w_S are subjective weights that determine the relative contribution of a species or species group *S* to the index, $N(v_{Sj})$ is the normalized value for a species *S* at shoreline segment *j*, and *T* equals 41 the total number of components (i.e., species or species groups) included in the assessment. The weights are normalized so that they sum to 1, and therefore, the index is effectively a weighted average. The resulting average score was renormalized within sub-basins so that the maximum value equaled 1.

All weights in equation 4.4 were equal. We could have assigned larger weights to species or habitats that we thought were more important, such as federally listed salmon species or eelgrass, but that involves making value judgments that we wished to avoid in this assessment. Such value judgments should be informed by the opinions of stakeholders and policy makers.

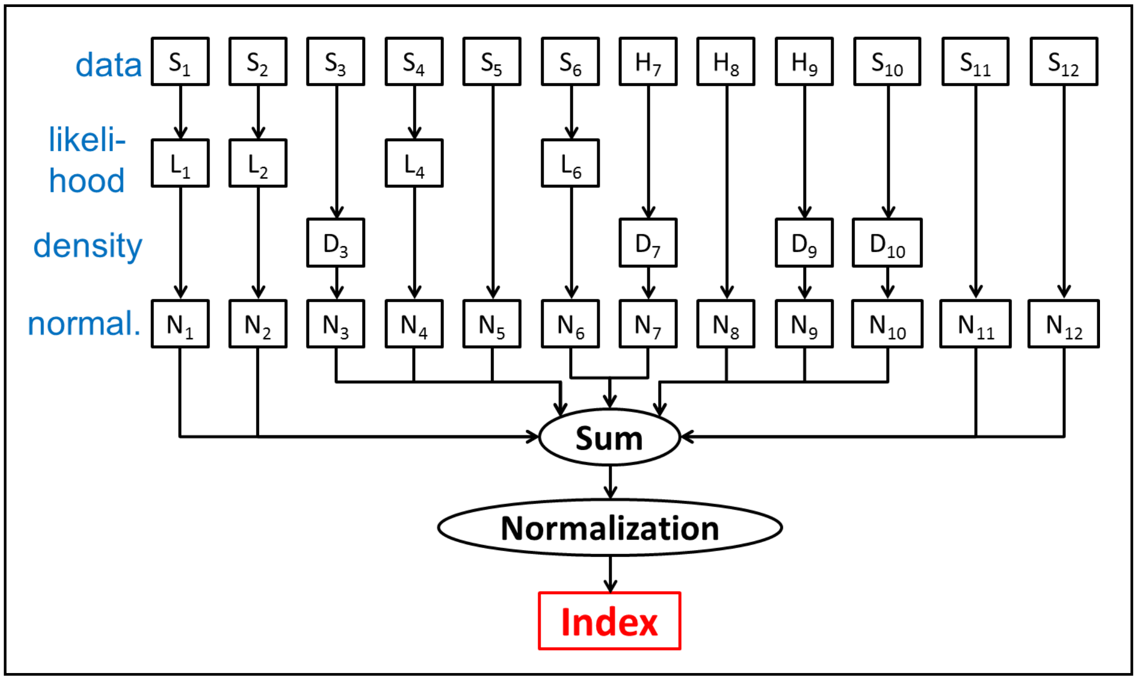


Figure 4.7. Process for calculating the composite index for a single shoreline segment. Using species (S) and habitat (H) data a relative likelihood of occurrence model (L) was generated for some species and a density (D) was calculated for other species and habitats. All values were normalized (N) from 0 to 1 relative to other segments in the same sub-basin. The normalized values are summed and then renormalized so that the maximum sum equals 1 within each sub-basin.

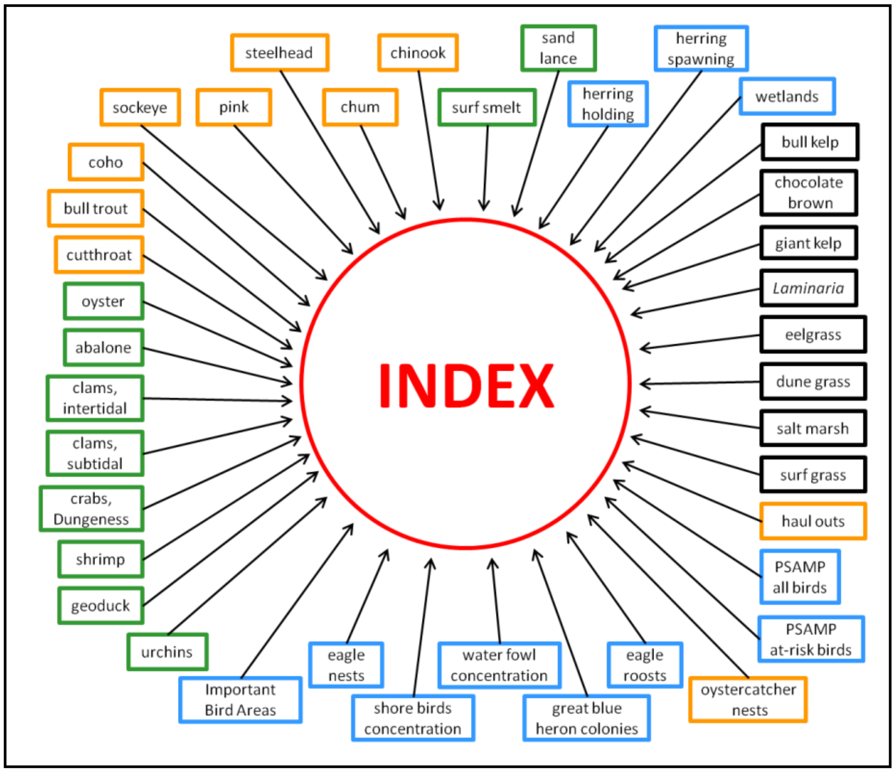


Figure 4.8. Data used in the calculation of conservation value indices. Forty-one types of data contributed to the indices. Salt marsh box includes sedges, high salt marsh, and low salt marsh (Table 4.3). Different data come in different forms: green = relative likelihood of occurrence models, orange = counts; blue = density; black = amount.

The average value produced by equation 4.4 can obscure sites that are relatively important for a single species or species group but relatively unimportant for most other species. Managers and planners need to be aware of such sites, and hence, a second index compensates for this shortcoming of the composite index. The second perspective, i.e., that value is best determined by a place's most significant contribution, can be implemented by taking the maximum value of $N(v_{sj})$ for each segment *j*. Because many of the data are presence/absence, 78% of shoreline segments had at least one $N(v_{sj})$ equal to 1. Hence, to obtain a more informative index that would more clearly discriminate among shoreline segments we chose to average the five largest $N(v_{sj})$ at each segment. The resulting score was renormalized so that the maximum value in each sub-basin equaled 1. We called this the *top-5 index*.

Given the many subjective judgments necessary to develop the index, we opted for parsimonious indices. One simplification was assigning equal influence to every component. We could have elicited expert opinions on the relative importance of each component, which would necessarily involve both technical and policy experts, but we did not. Another simplification was the index's structure. Our composite index had a flat structure (Figure 4.8). An alternative structure is hierarchical in which similar components are grouped together and the groups are assigned weights that determine their relative influence. In both instances, relative influence of components and the index structure, we chose to minimize subjective judgments, which led to equal influence and the flat structure.

Larger Grain Assessment Summaries

Our assessment units for the shoreline habitats assessment are the intersection of shoreform types and shorezone types. These spatial units represent real differences in geomorphology, topographic slope, and substrate – characteristics which we associate with shoreline habitat types. The units, which have an average length of 0.24 miles, are an appropriate grain for site-level and many local planning activities. However, for the purposes of guiding regional protection and restoration actions, our assessment units are too small. Larger grain assessment summaries were formed in two ways. First, we calculated the average composite index for PSNERP shoreline process units, which have an average length of 3.6 miles. Second, we calculated an average composite index for each shoreline segment using a 2-mile wide moving window which assigns to each shoreline segment the average value of all segments within 1 mile to either side of the segment.

Index Properties

We examined the index through various statistical and graphical analyses, evaluated the index's sensitivity to each component, and explored the potential effects of uncertainty on the index.

The index has 41 components. We examined the average contribution of each component to the composite index. The contribution of component *S* to the composite index at shoreline segment *j* is:

$$C_j = {w_S N(v_{Sj})} \big/ {AV_j} \tag{4.5}$$

and the average contribution of a component to the composite index is

$$\bar{C} = \frac{1}{M}\sum_{j=1}^{M} C_j \tag{4.6}$$

A sensitivity analysis was done to understand the relative influence of each component on the index. For the composite index, we defined sensitivity as:

$$C_j = {\Delta AV_j} \big/ {\Delta[w_S N(v_{sj})]} \tag{4.7}$$

Sensitivity analysis was done by calculating the composite index for all shoreline segments with the weights set to 1, recalculating the composite index after altering a single weight by a small amount (e.g., 5%), and applying equation 4.7 to each shoreline segment. The process was repeated for each component. The index score of each shoreline segment is effectively a separate model output, and hence each segment has its own sensitivity to each component. A mean sensitivity was calculated for each component by averaging over the separate sensitivities of all segments.

Uncertainty analysis was done by assigning a uniform or triangular probability distribution to each of the 41 weight parameters in equation 4.4. The distributions spanned the range of reasonable values for each parameter, but the mean and median of every distribution equaled 1. Parameter values were randomly selected from the 41 distributions and the index was recalculated for each shoreline segment. This was repeated for 200,000 iterations. The composite index score of each segment is effectively a separate model output, and hence each segment has its own distribution of index scores. The uncertainty associated with each segment's index score was represented as a histogram and the interval containing 90% of all index scores was determined for each segment.

4.3. Results

4.3.1. Index Components

Relative Likelihood of Occurrence

We constructed LO models for 10 species (Table D7). The models were simplistic, but nevertheless, the results generally conformed to our knowledge of these species' habitat associations (Simenstad et al. 1991, Dethier 2006). The relative likelihoods of occurrence for abalone and urchin (Figure 4.9), for instance, are highest on boulder, bedrock, and cobble substrates. The relative likelihoods of occurrence for surf smelt and sand lance are highest for sand and gravel and lowest for bedrock and boulder substrates.

Normalized Values

The composite index has 41 components and the properties of the components varied. For instance, commonness varied among components. For components without an LO model, the percent of shoreline segments with non-zero values reflects the commonness of that component in Puget Sound. For instance, eagle nests occur near 19% of segments but black oystercatchers nests occur on only 1% of segments (Figure 4.10). Components with an LO model had many more shoreline segments with non-zero values than components without LO models. According to the LO model for Dungeness crab, that species could occur (relative likelihood > 0) on 95% of shoreline segments. In contrast, bald eagle communal roosts, which did not have an LO model, occur near only 1% of segments.

The distributions of normalized values were also different among components. For instance, the distribution of normalized non-zero values for waterfowl concentration areas were uniformly distributed but normalized non-zero values for eagle nests were right-skewed, i.e., more segments with low values than high values (Figure 4.11). Also, normalized non-zero values for wetlands were approximately unimodal but normalized non-zero values for sedge were bimodal, i.e., high proportions of segments with high values or low values and lower proportion of segments with intermediate values.

Correlations among normalized values (Table D8) were mostly low, with 87% of correlations less than 0.2. Six percent of correlations among normalized values were moderate ($0.2 \leq \rho < 0.6$). This analysis indicates that nearly all components add unique information to the index. The highest correlations ($\rho >$ 0.75) were among salmon species because many species co-occur in the same streams.

Mean and median number of habitat functions (i.e., number of non-zero components) per shoreline segment were 10.5 and 11, respectively. The maximum number of habitat functions amongst all segments was 25. Correlation of the composite index score with number of habitat functions per segment was 0.56. Hence, the composite score was not based solely on the number of functions per segment. The quantity of certain functions in each segment, such as the area of eagle nest sites or the amount of eelgrass, affected the composite score.

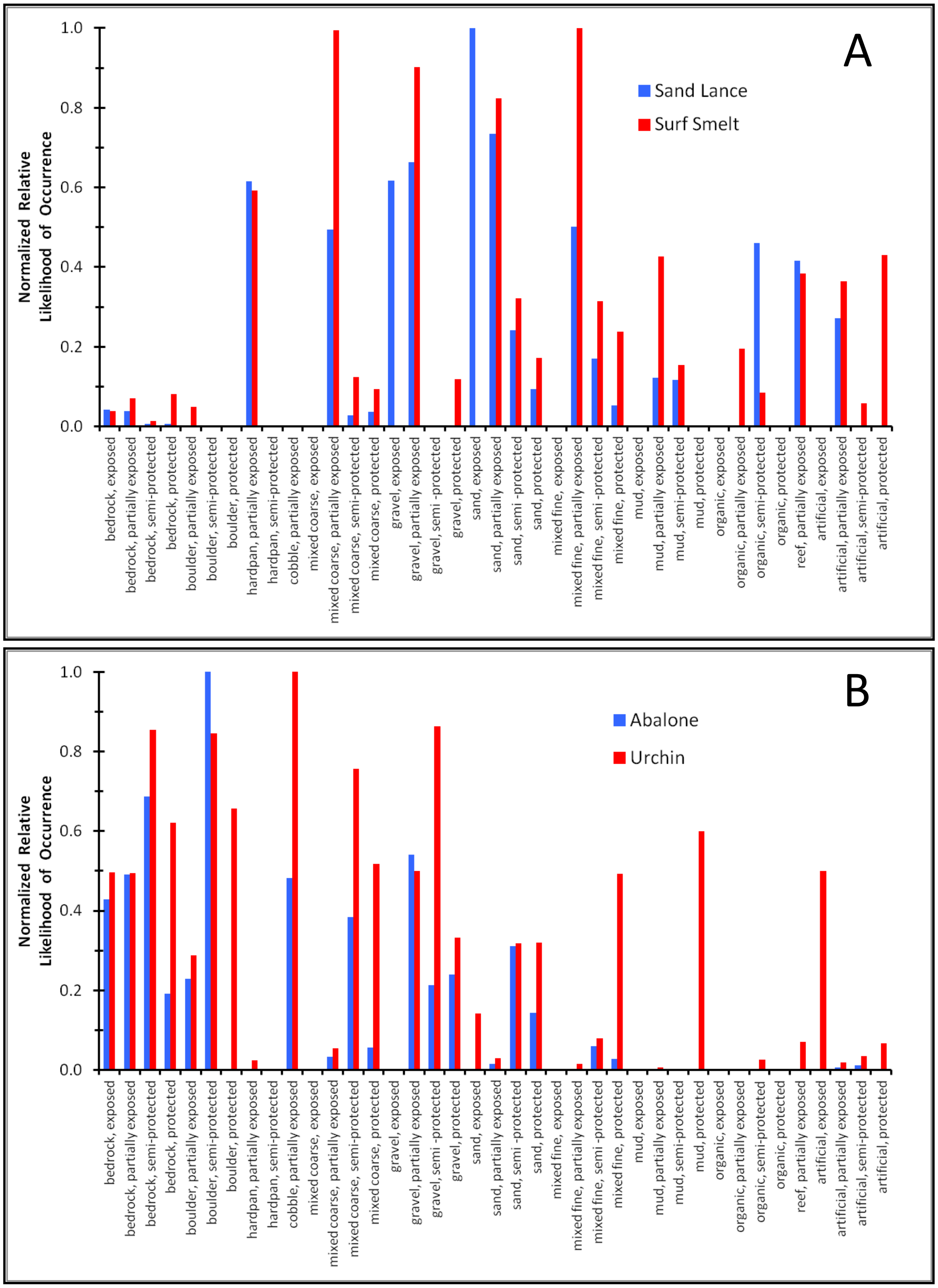


Figure 4.9. Results of relative likelihood of occurrence models for two species of forage fish which prefer sandy-gravelly substrates (A) and for two species, abalone and sea urchin, known to prefer rocky substrates (B).

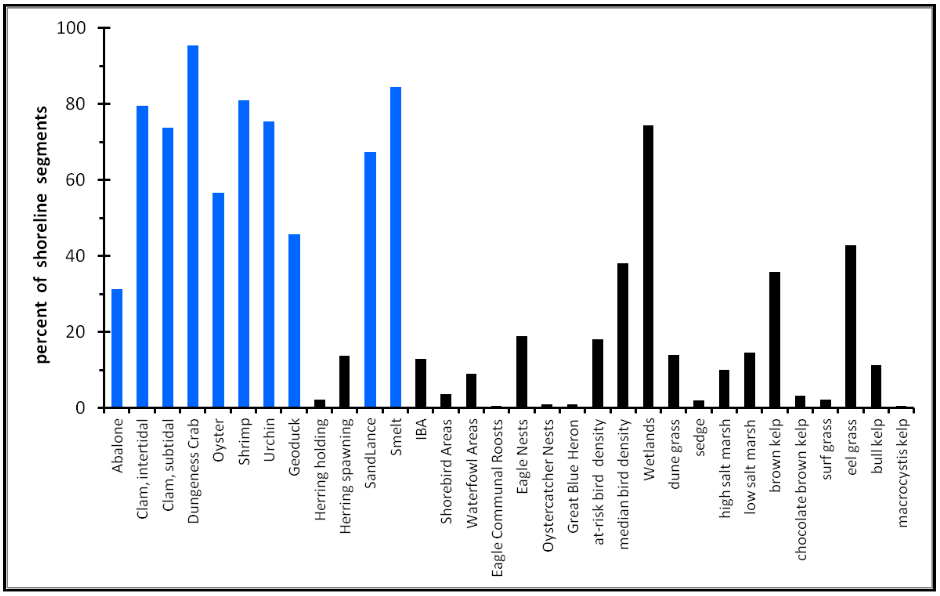


Figure 4.10. Percent of shoreline segments with non-zero values for each component of the composite index. Blue bars indicate components with a relative likelihood of occurrence (LO) model.

4.3.2. Indices of Relative Conservation Value

The basic results of the shoreline habitats assessment are maps (e.g., Figure 4.12). The map shows the relative value of every shoreline segment. In many places the index scores conform to our expectations. For instance, the relatively intact mouths of the Nisqually and Skokomish rivers have high index values and the degraded shorelines of Olympia and Shelton have low index scores. This pattern is repeated throughout the Puget Sound – the shorelines along large urban areas (Tacoma, Seattle, Bremerton, Everett) tend to have low scores and shorelines along areas known to have high ecological value (e.g., , Semiahmoo Spit, mouth of Dungeness River) have high scores. Two notable exceptions to these patterns are the mouths of the Puyallup and Duwammish Rivers. These river mouths are heavily degraded but the presence of many salmon species results in these shorelines having high relative value. Compared to the best shoreline segments in each sub-basin, the majority of segments have relatively low scores – 68 percent of shoreline segments had scores between 0.1 and 0.4.

Scores for the composite index have a right-skewed normal distribution (Figure 4.13) with a mean of 0.28 and about 1% of shoreline length being above 0.9 and 12% being below 0.1. The distribution of index scores varied by sub-basin (Figure 4.16). For instance, the distribution was skewed right in the San Juan Sub-basin with a mean of 0.15, but more symmetric for the Whidbey Sub-basin with a mean of 0.31.

The composite index and top-5 index were highly correlated, $\rho = 0.81$, but there are obvious differences between the two indices (Figures 4.12 and 4.14). The distribution of the composite index is highly skewed with only 11% of shoreline length having index values greater than 0.5. In contrast, the distribution for the top-5 index is more uniform with 49% of shoreline length having index values greater than 0.5. Also, almost 10% of shoreline length had scores greater than 0.9 for the top-5 index. Hence, the top-5 index indicates that a high proportion of shorelines length has moderate to high score for at least several components of the index.

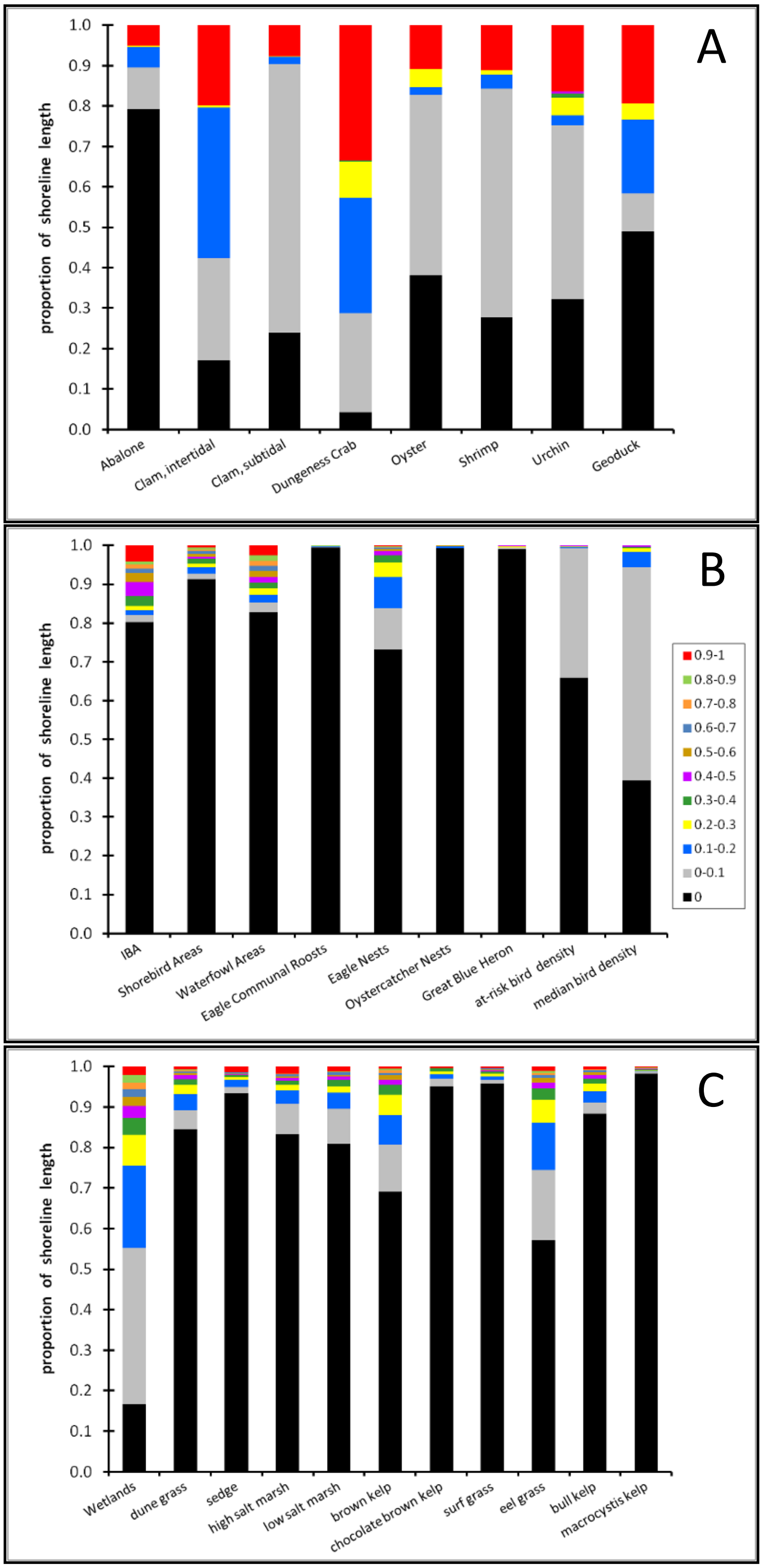


Figure 4.11. Distribution of normalized values for various index components: A) shellfish, B) birds, and C) wetlands and shoreline vegetation. Relative likelihood of occurrence models were used for all shellfish species. Zero value indicates the component is not present in the shoreline segment. Component values normalized from 0 to 1 within sub-basins.

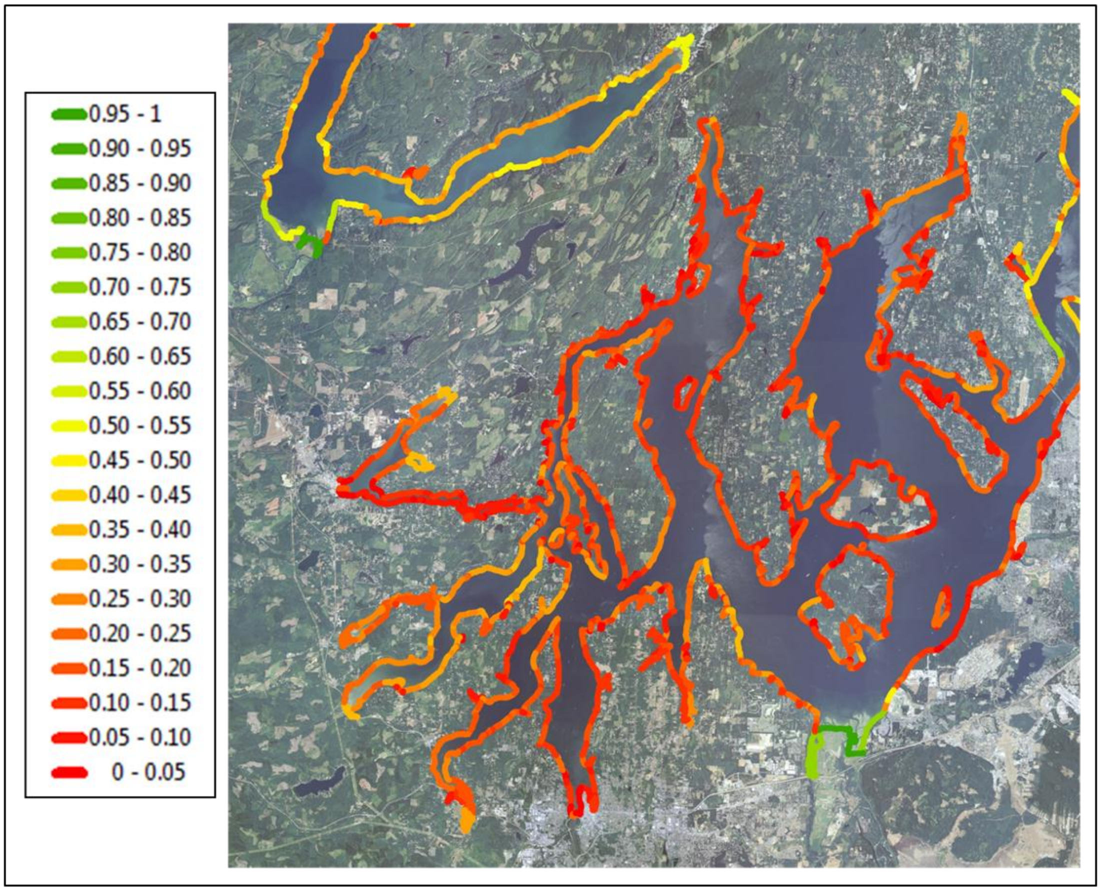


Figure 4.12. The Composite Index. The composite index is the normalized mean of all 41 components. Highest relative habitat value is dark green and lowest relative value is dark red. This example depicts South Sound Sub-basin and part of the Hood Canal Sub-basin (upper left hand corner). Index scores were normalized within oceanographic sub-basins.

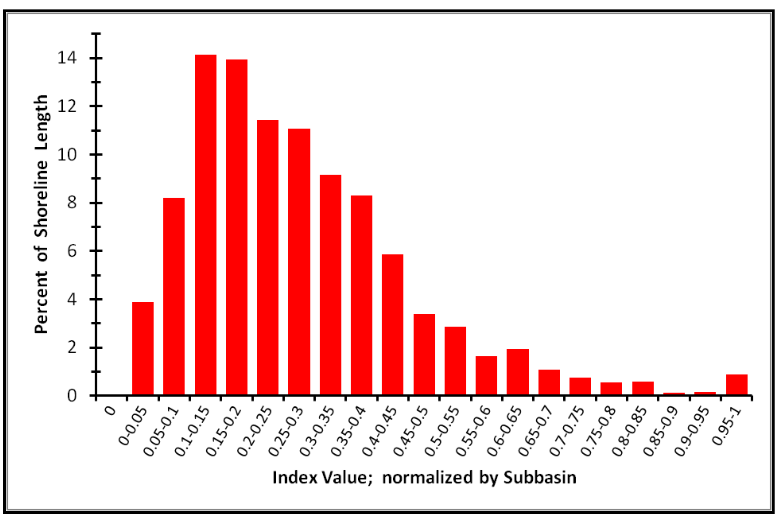


Figure 4.13. Distribution of composite index scores for all Puget Sound shorelines. Mean weighted by segment length equals 0.28.

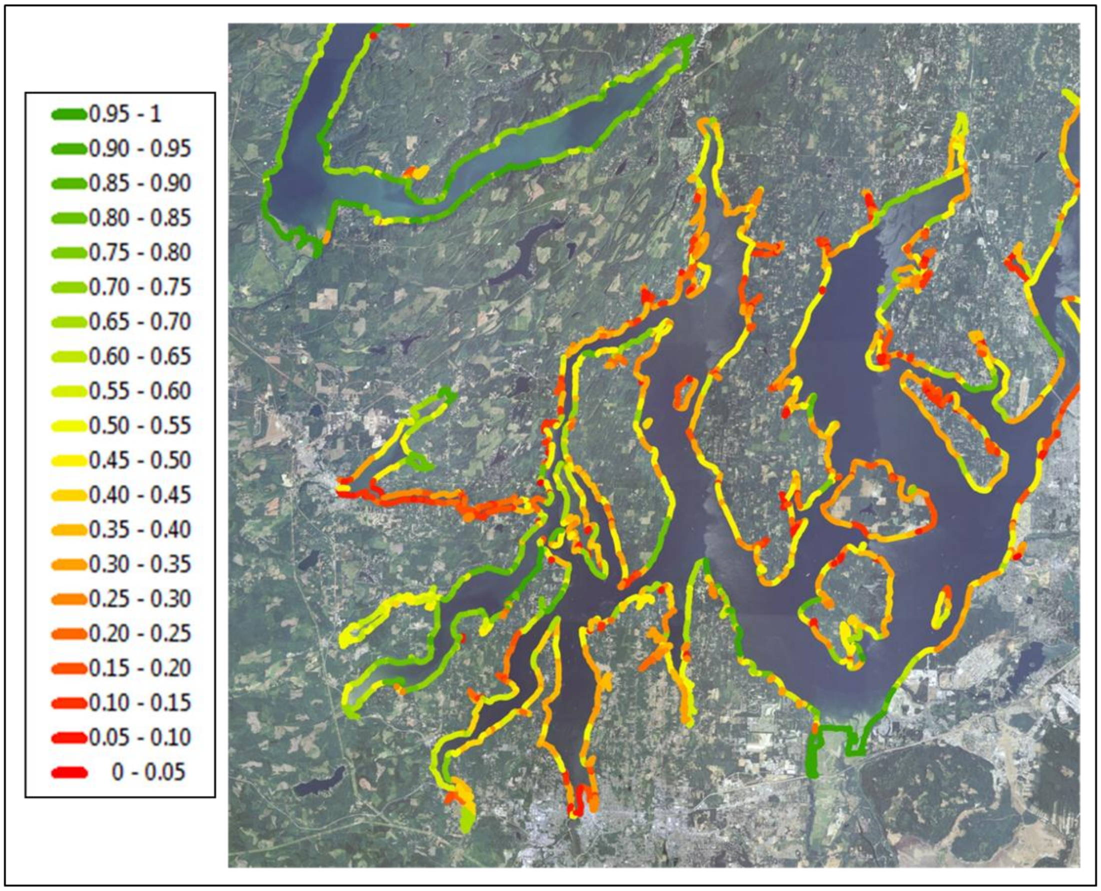


Figure 4.14. The top-5 Index. The top 5 index is the normalized mean of the five highest components in each shoreline segment. Highest relative habitat value is dark green and lowest relative value is dark red. This example depicts South Sound Sub-basin and part of the Hood Canal Sub-basin (upper left hand corner). Index scores were normalized within oceanographic sub-basins.

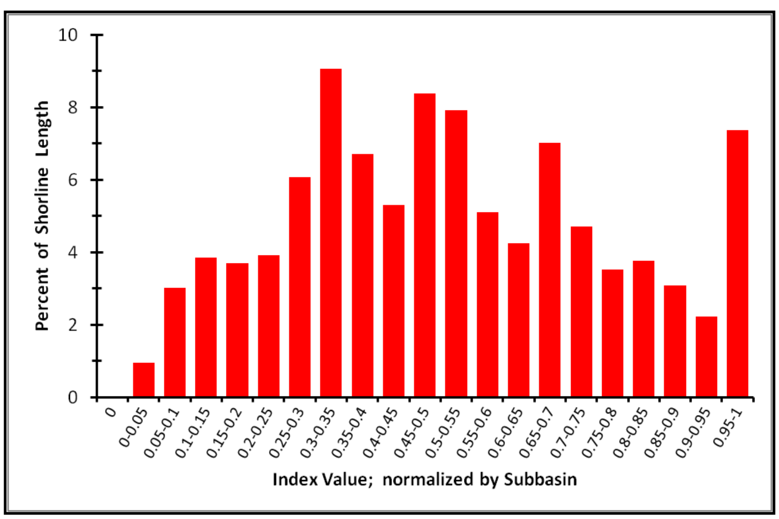


Figure 4.15 Distribution of top-5 index scores for all Puget Sound shorelines. Mean weighted by segment length equals 0.51.

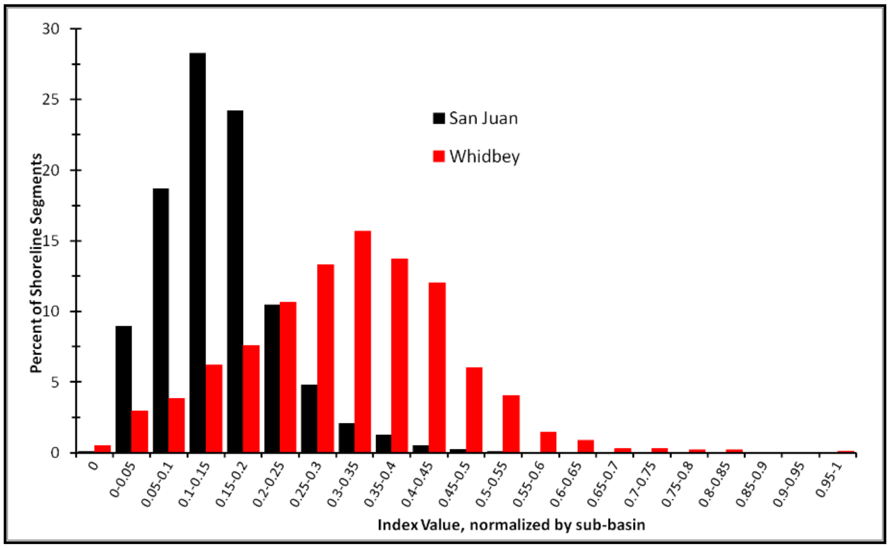


Figure 4.16. Distribution of composite index scores for the San Juan and Whidbey oceanographic sub-basins.

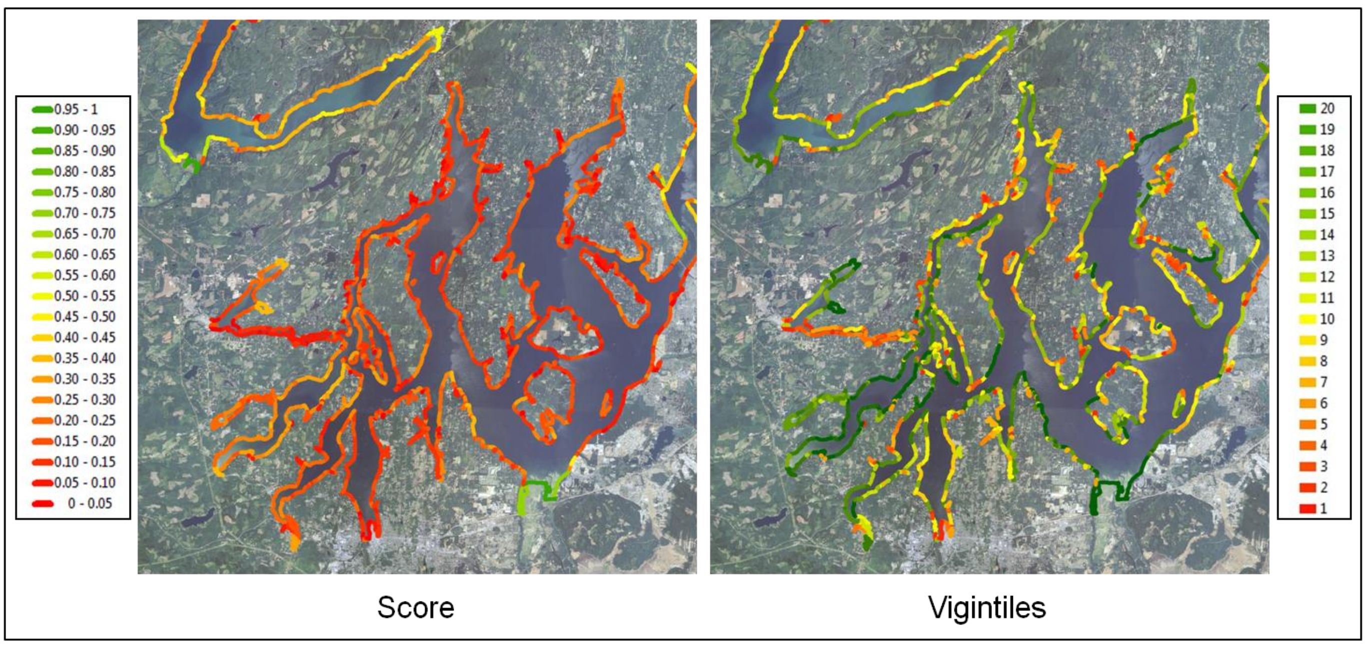


Figure 4.17. Comparison of scores and quantiles. For easier interpretation continuous index scores (left panel) were converted to 20 quantiles (i.e., vigintiles; right panel). Highest relative habitat value is dark green and lowest relative value is dark red. This example depicts South Sound Sub-basin and part of the Hood Canal Sub-basin (upper left hand corner of each map). Index scores were normalized within sub-basins.

The components most often comprising the top-5 index, in decreasing order of frequency, were: Dungeness crab, surf smelt, hardshell clams, urchin, wetlands, pandalid shrimp, geoduck, northern abalone, and Pacific oyster. For more than 20% of shoreline segments, at least one of these species or species groups contributed to the top-5 index. Dungeness crab contributed to the top-5 index for 81% of shoreline segments.

The previous results are for the magnitude of the index. In many practical applications we only need to know the rank of shoreline segments. That is, we only need to know which segments have higher or lower index scores than other segments. To examine ranks, we converted the continuous index score into 20 quantiles (i.e., vigintiles; Figure 4.17). Think of each quantile as a binning of ranks into categories. Each vigintile contains 5 percent of the shoreline length in its respective sub-basin. In other words, 5 percent of the shoreline will have the highest rank category and 5 percent will have the lowest rank category, and the remaining 90 percent of the shoreline is evenly distribution among the other 18 rank categories.

When comparing the composite index with the quantized composite index (Figure 4.17) we see that places with extreme low (Olympia, Shelton) or extreme high (mouths of Nisqually and Skokomish rivers) index scores are also in lowest and highest quantiles, respectively. Places with more moderate values for the composite index are distributed among the intermediate quantiles. As a result, some segments with relatively low index scores may have high ranks (i.e., are in a high quantile).

The assessment also enables us to compare the relative habitat value of shoreforms and habitat types. River deltas, barrier beaches, and bluff-backed beaches tend to have above average mean scores for the composite index (Figure 4.18). Artificial shoreforms had higher average scores than anticipated because many of them are associated with river mouths (i.e., Puyallup and Duwwamish Rivers) where all salmonid species are present. Pocket beaches, barrier lagoons, and open coastal inlets have below average mean scores, which is contrary to our intuitions about their habitat value. The pattern of mean relative habitat value for shoreforms was different among sub-basins. Mean scores exhibited little difference across shoreforms in the San Juan Sub-basin. In contrast, deltas had mean scores much greater than other shoreforms in the South Puget Sub-basin. Among the Dethier (1990) habitat types partially exposed types tended to have higher mean scores than semi-protected types (Figure 4.19). Habitat types with organic substrates tended to have much higher mean scores than other substrates. Organic substrates were often located at river mouths and river mouths have relatively higher index scores because all salmonid species are present at most major river mouths.

The larger grain assessments can guide regional protection and restoration actions. There was a high level of concurrence between the two larger grain assessments (Tables 4.4 and 4.5). In five of the seven oceanographic sub-basins, river mouths were among the top five places. This is especially true in the Juan de Fuca and Hood Canal sub-basins where river mouths are still relatively undeveloped. According to the moving window average, spits were another feature commonly among the top 5 places for most sub-basins.

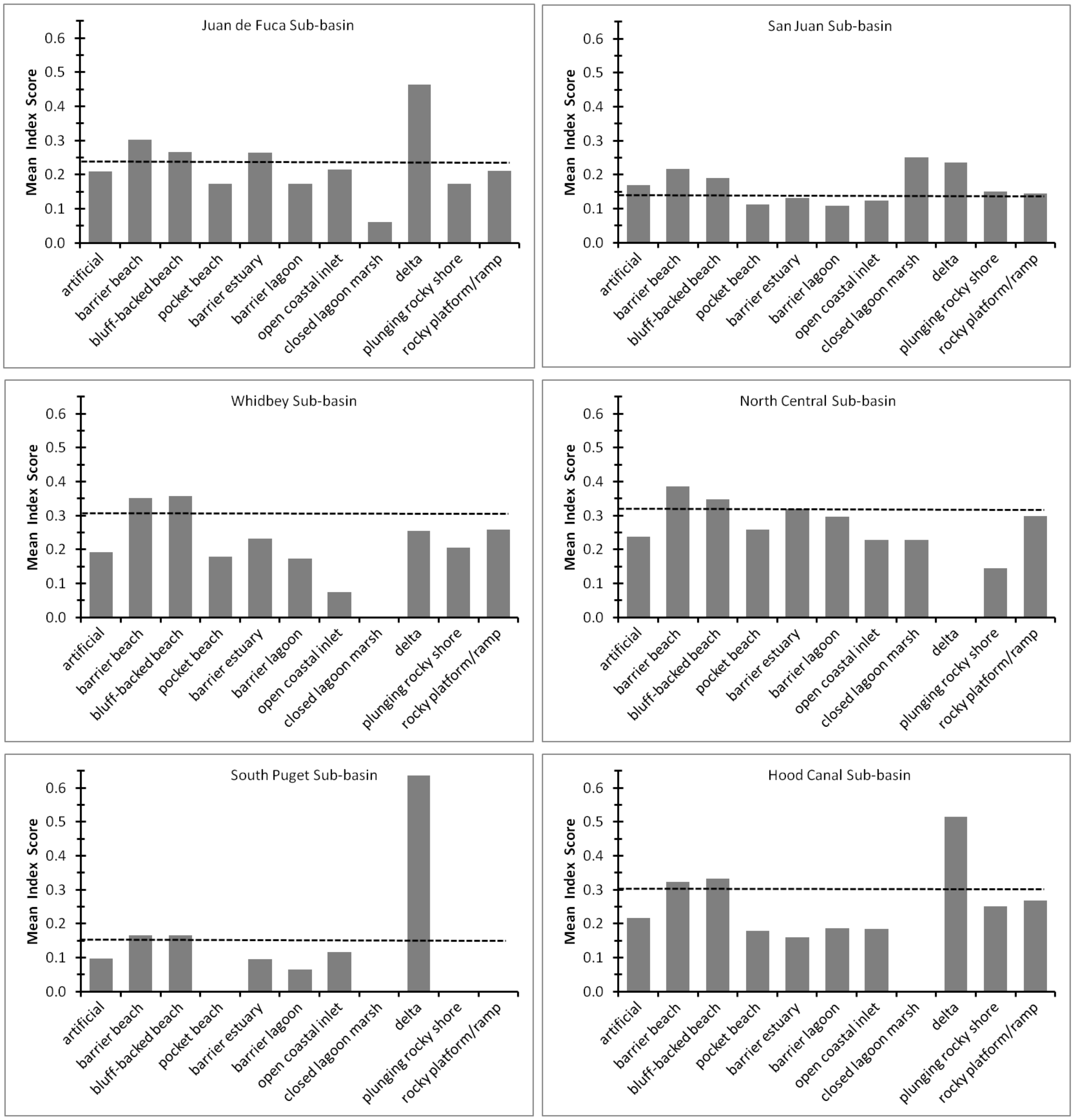


Figure 4.18. Mean composite index of relative habitat value for shoreforms for six of the seven sub-basins. Dashed line is average score (not weighted by segment lengths) for all shoreforms within each sub-basin. No bar means the habitat type does not exist in that sub-basin.

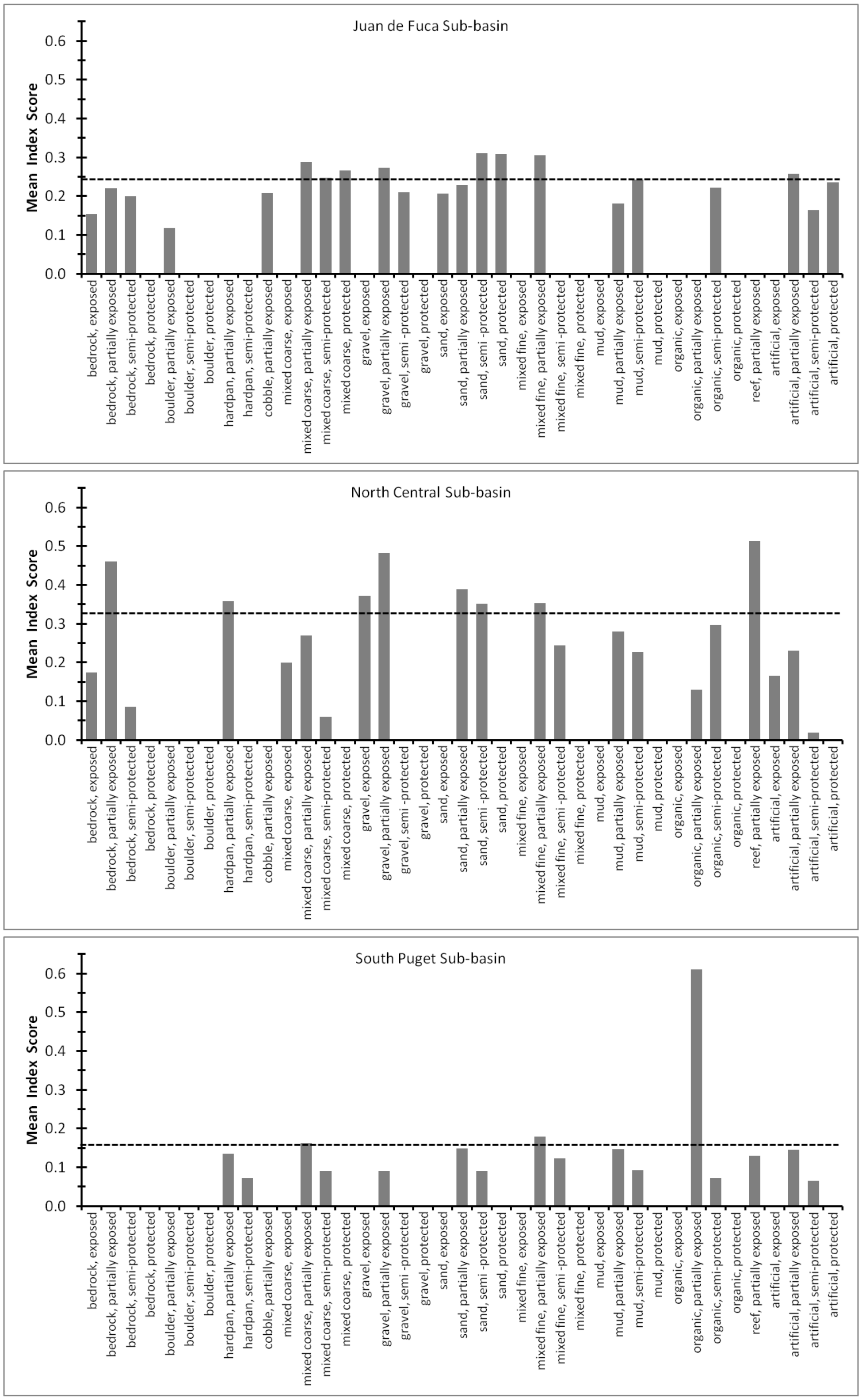


Figure 4.19. Mean composite index of relative habitat value for Dethier (1990) habitat types for three of seven sub-basins. Dashed line is average value (not weighted by segment lengths) for all habitat types within each sub-basin. No bar means the habitat type does not exist in that sub-basin.

Table 4.4. PSNERP shoreline process units with the highest mean composite index scores in each sub-basin. A succinct geographic description of each place is provided, but process unit identification numbers should be used for exact location.

Sub-basin	Process Unit ID	Geographic Description
Juan De Fuca	• 9016	mouth of Dungeness to Kulakala Point
	• 1028	from Slip Point almost to Pillar Point
	• 1023	inside of Dungeness Spit
	• 9017	mouth of Elwha
	• 1024	inside Dungeness Spit
San Juan	• 9002	mouth of Nooksack
	• 7166	northeast side of March Point
	• 7167	north end of March Point
	• 7143	Birch Point to Semiahmoo Spit
	• 7168	Crandall Spit, north end of March Point
North Central	• 5015	Indian Island, northeast shore
	• 5016	Indian Island, north shore
	• 5036	Indian Island, south tip and southeast shore
	• 5017	Indian Island, north shore
	• 5026	Kuhn Spit to Old Fort Townsend in Port Townsend Bay
Whidbey	• 6057	Gedney Island, north end
	• 6027	Ben Ure spit and south of spit
	• 6046	Camano Head, west side
	• 6060	Gedney Island, northeast side
	• 6051	Sundins and Jupiter Beaches near mouth of Stillaguamish River
Hood Canal	• 9014	mouth of Dosewallips River
	• 9011	mouth of Skokomish River
	• 9015	Quilcene Bay, from Indian George Creek to Camp Discovery
	• 2047	mouth of Duckabush River
	• 2028	Twanoh State Park
South Central	• 4065	Burke Bay
	• 4148	mouth of Duwammish River
	• 4128	Agate Point
	• 4147	Agate Pass, east shore
	• 4064	Burke Bay
South Puget	• 9009	mouth of Nisqually River
	• 3065	southeast shoreline of Totten Inlet
	• 3013	north of Dogfish Bight; south shore of Nisqually Reach
	• 3094	Chapman Cove off Oakland Bay
	• 3067	just north of Gallagher Cove in Totten Inlet

Table 4.5. Shorelines with the highest average composite index score in each sub-basin based on 2 mile moving window average. Where shorelines did not have an official place name we succinctly describe the location.

Sub-basin	Places with Highest Index Value
Juan De Fuca	• mouth of Dungeness River • from Slip Point to almost Pillar Point • Dungeness Spit • Travis Spit at Sequim Bay • mouth of Elwha
San Juan	• mouth of Nooksack River • Semiahmoo Spit • east side of March Point • island between Samish River and Edison Slough • Crandall Spit in Fidalgo Bay
North Central	• north and northeast shores of Indian Island • Harrowstone Island, north and south of Mystery Bay • Indian Island, south of Bishops Point • Indian Island, embayment wetland north shore of Oak Bay • isthmus between Indian and Harrowstone islands
Whidbey	• mouth of Snohomish • mouth of Stillaguamish • mouth of Skagit • south of Ben Ure Spit • south end of Camano Island
Hood Canal	• mouth of Skokomish River • mouth of Dosewallips River • mouths of Big and Little Quilcene Rivers • mouth of Duckabush River • east shore of Dabob Bay, from Camp Discovery to 2 miles south
South Central	• Burke Bay • near Old Man House State Park (Agate Pass) • north of Agate Pass Bridge on Bainbridge Island (Agate Pass) • mouth of Puyallup River • mouth of Duwammish River
South Puget	• mouth of Nisqually River • shoreline north of Nisqually River estuary • Kennedy Creek estuary • mouth of Deer Creek at head of Oakland Bay • south shore of Totten Inlet, near its mouth

4.3.3. Index Properties

Sound wide, the components that made the largest contribution on average to the composite index were Dungeness crab, urchins, surf smelt, intertidal clams, and wetlands (Figure 4.21). Twenty-four of the 41 components each comprised, on average, ≤ 1% of the composite index , and of those, 3 components each comprised ≤ 0.1% of the composite index on average. However, the components with small average contributions (≤ 1%) still made contributions ranging from 24 to 100% at individual segments. The components' contributions varied by sub-basin (Figure 4.22). For instance, the biggest contributors in the Juan de Fuca Sub-basin were urchins, Dungeness crab, and important bird areas (IBA). In contrast, the biggest contributions to the composite index in the South Puget Sub-basin were intertidal clams, geoduck, and surf smelt.

The sensitivity analysis showed that the composite index is most sensitive to the components that have LO models (Figure 4.23). This sensitivity is caused by the high proportion of shoreline segments that have non-zero values for LO models. The model output was also sensitive to the salmonid species data and had about the same degree of sensitivity to all 8 salmonid species. The sensitivity of the index to a component was moderately correlated ($\rho = 0.49$) with the component's average contribution to the index.

The uncertainty analysis indicates that a large proportion of shoreline segments with different index values may not be significantly different. That is, given the assumptions of our simple uncertainty analysis, the relative habitat value of many shoreline segments is effectively the same. In the Juan de Fuca Sub-basin, for example, the 200 shoreline segments with average index scores between approximately 0.2 and 0.4, are not significantly different from each other (Figure 4.24). How confident we can be about identifying the highest value segments differs amongst sub-basins. For instance, in the Hood Canal Sub-basin, the top two segments are significantly different than nearly all other segments in the top 40 (Figure 4.25). In contrast, the top two segments in the South Central Sub-basin are not significantly different than any other segments in the top 40.

Greater uncertainty does not mean that a shoreline segment has lesser value than that estimated by this assessment. Greater uncertainty means that the actual relative habitat value could be larger or smaller than the estimated value.

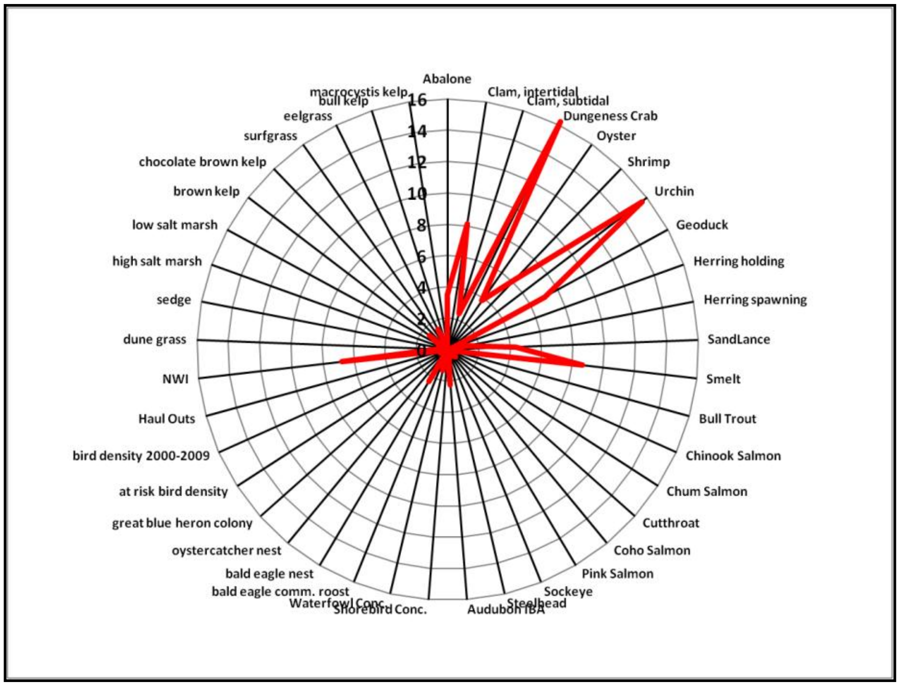


Figure 4.21. Mean percent contribution of components to composite index of relative habitat value. There is a radial axis for each of the 41 components of the composite index. NWI and IBA mean national wetlands inventory and important bird areas, respectively.

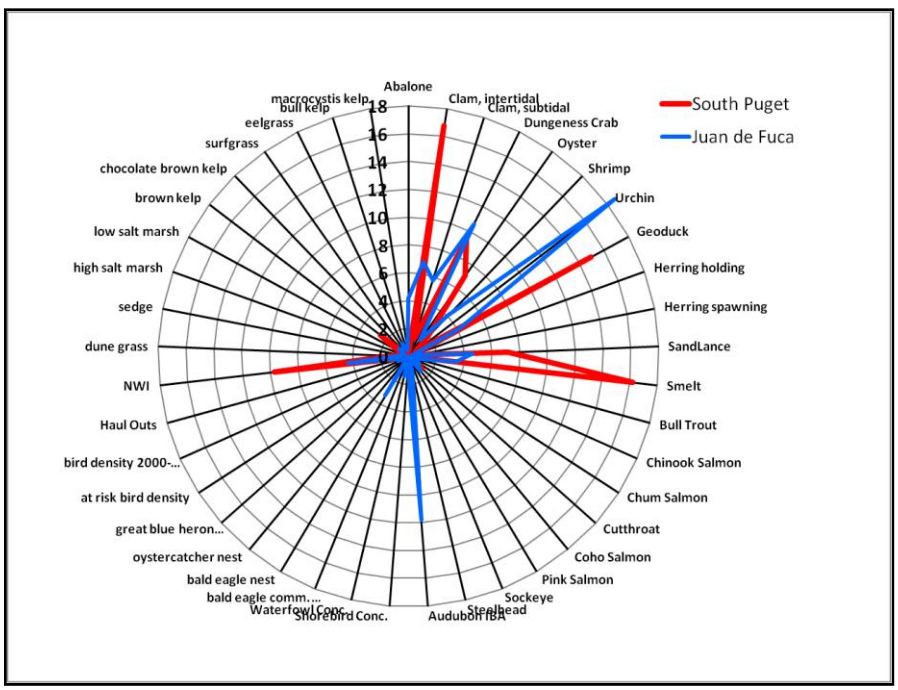


Figure 4.22. Mean percent contribution of components to composite index of relative habitat value for South Puget and Juan de Fuca sub-basins. There is a radial axis for each of the 41 components of the composite index. NWI and IBA mean national wetlands inventory and important bird areas, respectively.

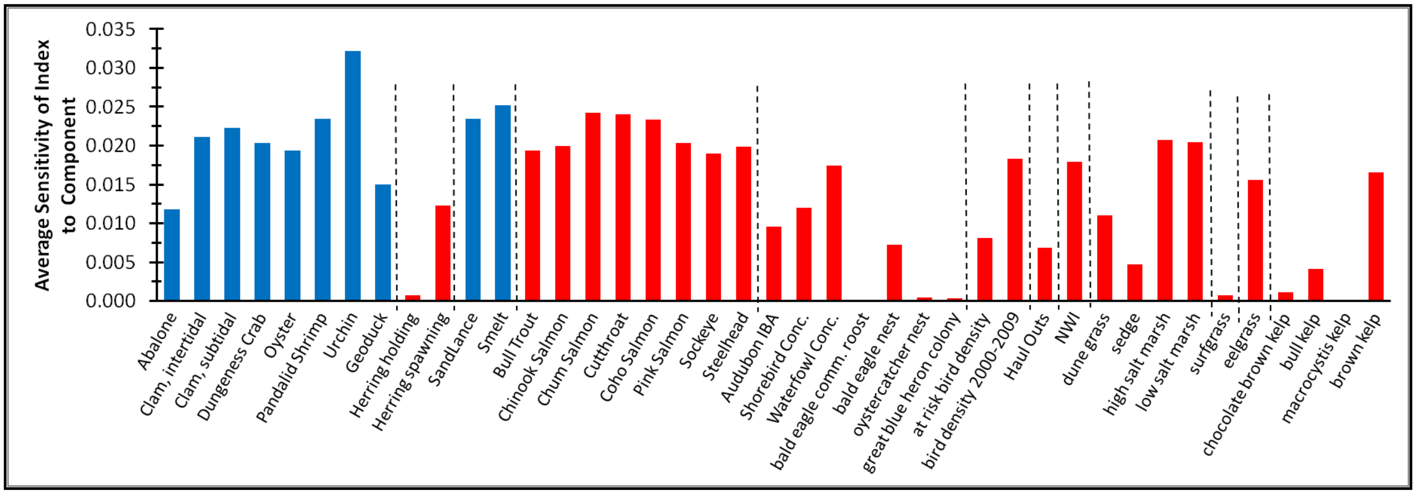


Figure 4.23. Sensitivity of the composite index to each of the components. Blue bars correspond to species with relative likelihood of occurrence (LO) models. Vertical dashed lines demark groups with similar taxonomy or functional characteristics.

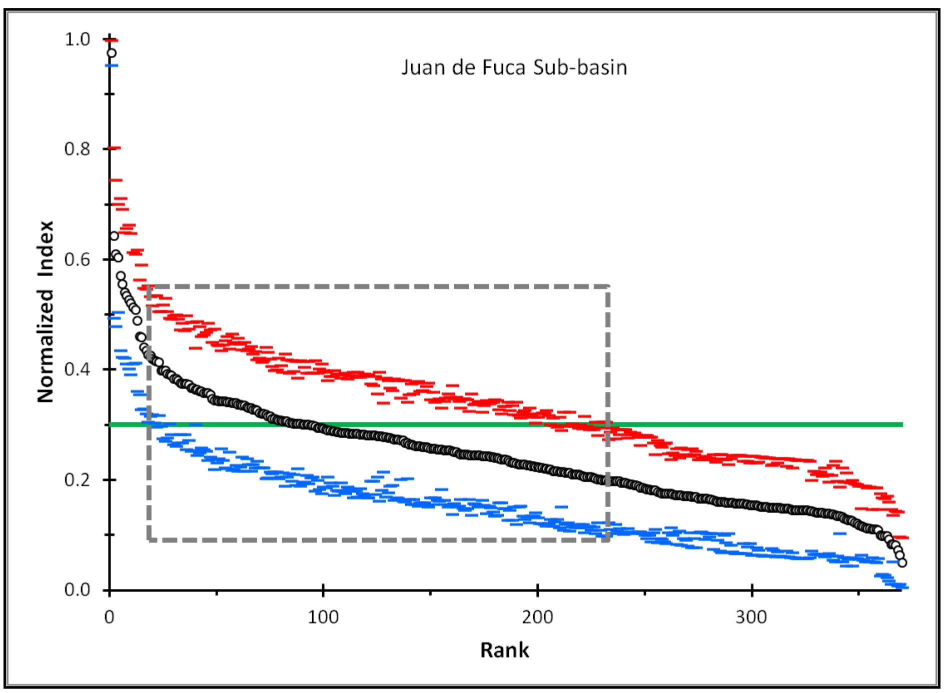


Figure 4.24. Results of the uncertainty analysis for the Juan de Fuca Sub-basin. The 372 shoreline segments in the Juan de Fuca Sub-basin are sorted from highest to lowest average composite index score. Hollow circles are average scores for each shoreline segment. Red and blue bars encompass 90% of simulation replicates. That is, the bars demark the 90% confidence interval. Green line arbitrarily set at average index = 0.3. Shoreline segments within dashed box are not significantly different according to the uncertain analysis. Segments to the left and right of the dashed box are significantly different.

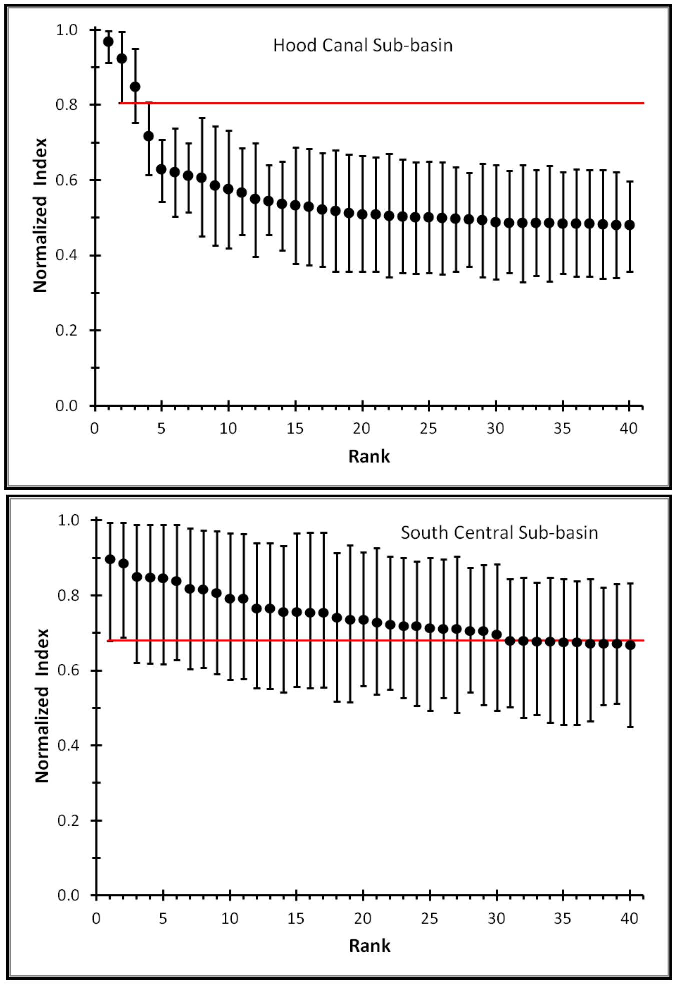


Figure 4.25. Examples of results from uncertainty analysis by sub-basin. Shoreline segments are sorted from highest to lowest component index score. Only top 40 shoreline segments in sub-basin are shown. Filled circles are average score for each shoreline segment. Error bars encompass 90% of simulation replicates. That is, the bars demark the 90% confidence interval. Red line delineates location of lower limit for particular segment: second best shoreline segment in Hood Canal and best shoreline segment in South Central. Upper limits below the red line indicate a significant difference.

4.4. Discussion

The main products of the marine shoreline habitats assessment are two indices that summarize disparate data on the occurrence or abundance of 41 species, species groups, and habitat types. The indices indicate the *relative value* of marine shoreline segments based on habitat functions – i.e., higher scores indicate shoreline segments with relatively more habitat functions than segments with lower scores. We developed two indices because relying on a single index would obscure important information. The composite index is a sum of 41 components, hence, it mainly reflects the quantity of habitat functions at shoreline segments. In general, shoreline segments with many habitat functions obtain the highest scores for the composite index. However, if we relied on only the composite index, then segments with a small number of habitat functions but high relative value for only a small number of functions would obtain low scores. We want to know about such places – i.e., those that have high value for a few habitat functions. The top-5 index was created to provide that information and should be used in conjunction with the composite index. A comparison of Figures 4.12 and 4.14 demonstrates the necessity of the top-5 index. The relative value of many shoreline segments is much greater in Figure 4.14 than Figure 4.12.

The intended application of the marine shoreline habitats assessment is land use plans, such as shoreline master programs, produced by local governments. City and county governments have regulatory authority over land use along marine shorelines, and the allowed land uses they designate through shoreline master programs and comprehensive plans may be the most important actions affecting the health of marine shoreline habitats. Local land use planning along marine shorelines is governed by Washington's Shoreline Management Act (SMA; RCW 90.58). The governing principles (WAC 173-26-186) of the shoreline guidelines (WAC 173-26-176) established under the SMA state, "Local [shoreline] master programs shall include policies and regulations designed to achieve no net loss of those ecological functions." Any shoreline segment with composite index score greater than zero contains or is in close proximity to at least one ecological function. According to our assessment, nearly every shoreline segment in Puget Sound (>99%), even highly degraded shorelines, has a composite index score greater than zero. When developing or revising land use plans, local governments can use this assessment to summarize the relative value of shoreline segments based on habitat functions and to create a partial accounting of habitat functions along all shorelines under their jurisdiction.

The intended application of the marine shoreline habitats assessment is quite different than the intended application of the terrestrial and freshwater habitats assessments. The main application of the terrestrial and freshwater assessments is local land use planning which is governed by Washington's Growth Management Act (GMA). Under GMA, local land use plans must accommodate projected human population growth (RCW 36.70A.110, 36.70A.115). The terrestrial and freshwater habitats assessments should be used to direct new growth away from places with relatively high habitat value and toward places with relatively low habitat value. In contrast, the SMA does not require the accommodation of human population growth, and regulations promulgated under the SMA stipulate no net loss of ecological functions. Therefore, the marine shoreline habitats assessment should *not* be used to direct new growth toward places with relatively low habitat value. The marine shoreline assessment should be used in conjunction with other information, such as PSNERP'S assessment (Cereghino et al. 2012), to direct restoration activities toward places with relatively low habitat value and to direct protection activities toward places with relatively high habitat value.

Further discussion of the assessments is provided in Part 5 of this report.

4.4.1. Validation

Validation entailed comparing the index scores against our collective knowledge of the Basin − does the index show places we believe to be relatively more important as more important and places we believe to be relatively less important as less important. In most places the index scores conform to our expectations. For instance, the relatively intact mouths of the Nisqually and Skokomish rivers have high index scores and the degraded shorelines of Olympia and Shelton, which support few habitat functions, have low index scores. This pattern is repeated throughout Puget Sound. Surprisingly, however, even some highly degraded shorelines obtained high relative values. Because eight migrating salmonid species are present there, the mouths of the Dumwamish and Puyallup rivers were amongst the highest scoring shorelines in their sub-basins. Because they are highly degraded, PSNERP's assessment (Cereghino et al. 2012) did not recommend the mouths of the Dumwamish or Puyallup rivers for protection or restoration. Nevertheless, planners should remain aware of the habitat functions at these and other degraded sites.

Comparing the results to our expectations revealed a potential shortcoming the assessment. The low mean scores for lagoons and inlets is contrary to our intuition about their habitat value. This may be due to the shortcomings of our data. For instance, our data on salmonid species presence are based mostly on adult presence. Other life stages are poorly represented in the data. Many coastal inlets are associated with small streams that do not contain adult salmon species but may support juveniles (E. Beamer, Skagit River System Cooperative, pers. com.). Furthermore, lagoons and inlets may provide sheltered foraging or resting areas for waterfowl but PSAMP bird surveys do not cover small inlets, and therefore, the presence of waterfowl or other water-dependent birds is unobserved and unrecorded. In short, the data available to us may underestimate the habitat value of lagoons and inlets.

Another form of validation is comparison to other ecological assessments of Puget Sound. During the past decade, two major efforts have published maps depicting "high priority" sites for marine habitat conservation in Puget Sound: the Washington Department of Natural Resources' priority marine sites (Palazzi and Bloch 2006) and The Nature Conservancy's nearshore conservation portfolio (Floberg et al. 2004). Both efforts were done for different purposes and used different methods. Unlike our assessment, these two other efforts did not assign scores to all shorelines. Instead, they identified a subset of shorelines that they determined to be the highest priority for conservation – Palazzi and Block (2006) identified 34 large sites and Floberg et al (2004) selected about 30% of all shorelines in Puget Sound. Palazzi and Block (2006) relied almost exclusively on expert opinion and Floberg et al (2004) used an optimization algorithm that found the most efficient set of sites for conservation. Palazzi and Block (2006) used criteria such as unusual spawning, nursery, or feeding areas; areas that include entire life history of a species; and areas that contain viable populations for which there are no empirical data; and included criteria such as adjacent to upland conservation areas, high ecological quality, and ecological connectivity which we did not include in our index. An influential factor in Floberg et al. (2004) was a cost index that directed site selection away from shorelines that were highly degraded. Given these differences between our assessment and these other assessments we expect differences in the results, Nevertheless, we also expect some congruence between our assessment's highest scoring shoreline segments and the high value sites identified by Palazzi and Block (2006) and Floberg et al (2004).

In comparing the results of our assessment to those of Palazzi and Bloch (2006) there are obvious similarities and differences (Figure 4.26). Our assessment and theirs identify the shoreline from Slip Point to Pillar Point, Kilisut Harbor, mouth of the Elwah River, mouth of Skagit River, mouth of the Nisqually River, and the Agate Pass area as conservation priorities. Our assessment assigned high index scores to the Dungeness Spit, Semiahmoo Spit, Ben Ure Spit, shorelines in Dabob Bay, mouths of Big and Little Quilcene Rivers, and the mouth of Kennedy Creek but Palazzi and Bloch (2006) did not identify these as conservation priorities. Other major differences in methods that further explain the differences in results are: 1) spatial scale – Palazzi and Block's high priority sites are many times larger than our assessment units; and 2) their assessment included all waters in Puget Sound, whereas our assessment covers only waters within 400 m of shorelines.

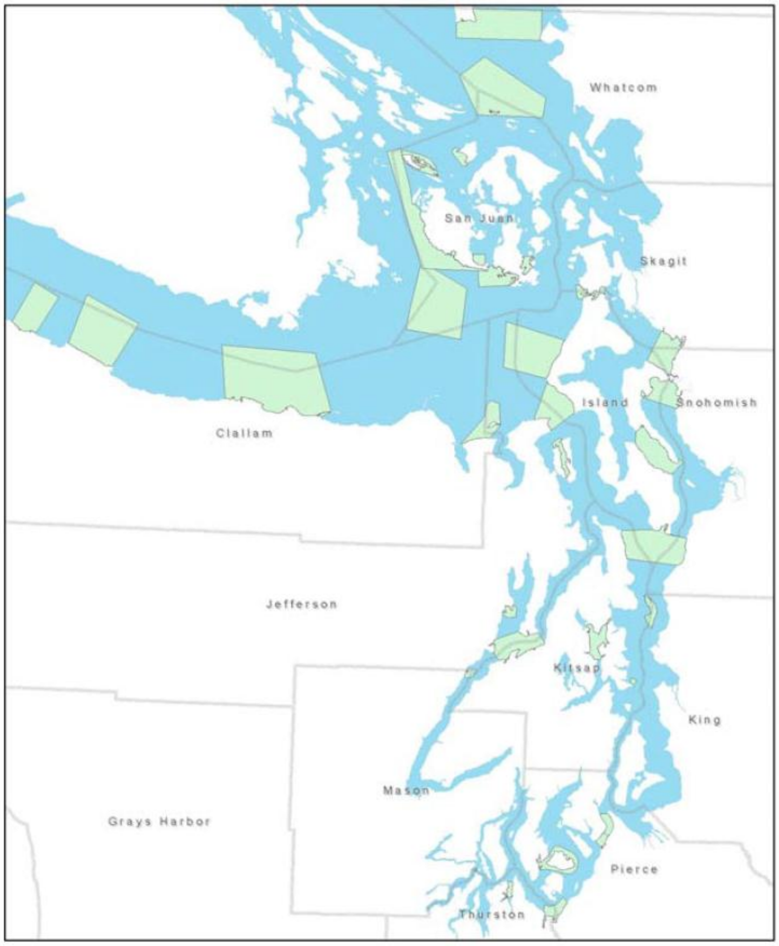


Figure 4.26. Priority marine sites for conservation in Puget Sound from Palazzi and Bloch (2006). Thirty four sites are shown in green.

Floberg et al. (2004) and our assessment both identify Sequim bay, Discovery Bay, Kilisut Harbor, Quilcene bay, Tarboo Bay, Agate Pass, Lynch Cove, mouths of the Skokomish and Nisqually Rivers, and mouths of Kennedy and Skookum creeks as high value sites (Figure 4.27). Floberg et al. (2004) did not identify the mouths of the Duwamish, Puyallup, and Snohomish Rivers as high value because those site have high costs for conservation. Other discrepancies between our assessment and Floberg et al. (2004) in the South Central and Whidbey sub-basins can be attributed to their cost index. Another major difference in methods that further explains the differences in results is that Floberg et al. (2004) did not divide Puget Sound into sub-basins, and therefore, their high values sites are unevenly distributed across Puget Sound and biased toward less developed portions of Puget Sound. In fact, very little shoreline in the Whidbey and South Central sub-basins were selected by Floberg et al. (2004), and this was due largely to the uneven distribution of their cost index across the seven sub-basins.

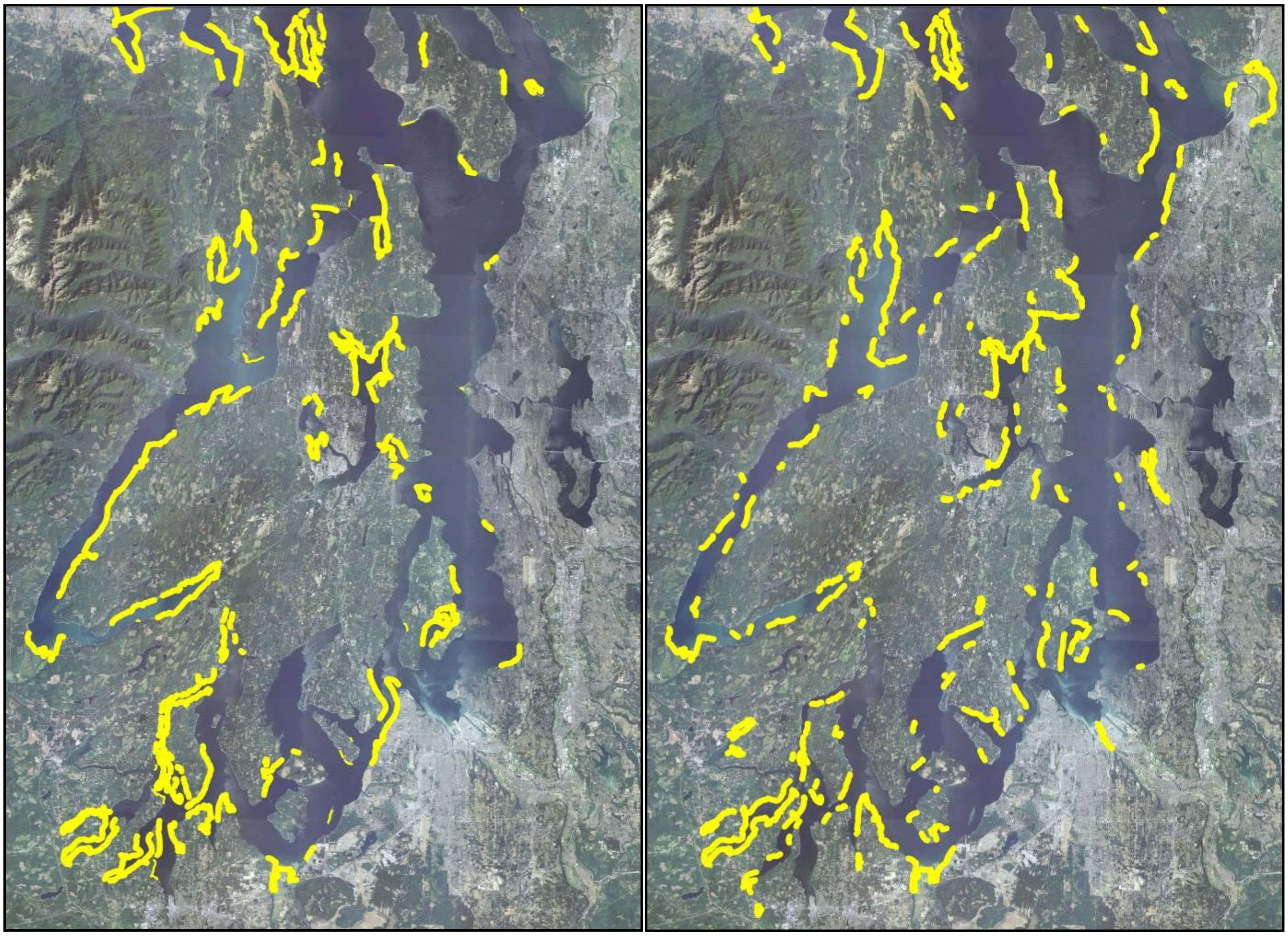


Figure 4.27. Comparison of The Nature Conservancy's nearshore conservation portfolio (Floberg et al. 2004; left panel) to our assessment (right panel). For our assessment, yellow denotes segments in the top 30% of composite index scores.

The differences between the results of this assessment and those of Palazzi and Block (2006) and Floberg et al (2004) demonstrate the folly in relying on a single assessment for planning and decision making. The three assessments serve different purposes and all three provide useful information.

4.4.2. Caveats

Our indices indicate the amount of co-occurring habitat functions for those particular species and life stages for which we collect data. In general, we collect occurrence data for particular species or habitat types because: 1) they are harvested; 2) they are rare or imperiled; or 3) they are highly sensitive to human disturbance. In other words, we collect data on those species and habitat types we are most concerned about. Therefore, we assumed that an assessment based on these data should indicate those places we should be most concerned about for the conservation of fish and wildlife. However, the occurrences of the species we used in this assessment very likely correlate with the occurrences of many other species. For instance, urchin and northern abalone, which were components of our index, are likely to co-occur with other species associated with rocky substrates. Nevertheless, the 41 components that comprise the indices are not a comprehensive accounting of the habitat functions for the many animal and plant species that are found along the marine shorelines of Puget Sound. We lack occurrence data for the majority of species. Other vital habitat functions, such as nearshore rearing habitat for juvenile salmonids, are not adequately addressed by our assessment.

The extent of our analysis covers the entire Puget Sound, however, we split the Sound into oceanographic sub-basins. The seven oceanographic sub-basins are based on bathymetry and circulation patterns, but these sub-basins also reflect regional patterns in shoreline geology, geomorphology, and wave environment (Shipman 2008), which manifest regional patterns in biological communities. We minimized comparisons of dissimilar biological communities by doing our assessment calculations within sub-basins. *Index scores cannot be compared across sub-basins*. Our assessment in its current form does not enable Basin-wide comparisons. Regional authorities should keep that in mind when using this assessment. The assessment may be useful for establishing regional priorities for protection and restoration of shorelines, but be aware that comparisons are only valid within sub-basins.

For most components that comprise the indices, we believed error rates in the occurrence data were acceptable. For a subset of components we believed error rates were likely to be unacceptable, and for these we developed LO models. LO models may overestimate the relative likelihood of occurrence, but we believed using a model with a high rate of false positives was more precautionary, and hence, preferable, to using data with a high rate of false negatives. Those components for which we did not develop an LO model were assumed to be equally accurate. This is unlikely to be true. Some datasets are regularly updated through annual systematic surveys (e.g., PSAMP bird surveys), while other datasets rely on the reporting and recording of incidental observations (e.g., bald eagle nests, great blue heron colonies). We could have compensated for these differences in accuracy by weighting some data sets more heavily than others, but we chose not do this because evaluating relative accuracy and assigning weights would entail numerous subjective judgments.

4.4.3. Potential Improvements

Our indices could be improved several ways. First and foremost, more up-to-date and accurate occurrence data are needed. Some of the occurrence data, in particular, those data described in WDF (1992), are decades old. The sensitivity analysis showed that the index is most sensitive to the LO models which were developed primarily for those shellfish species found in WDF (1992). Because we lack actual absence data, the accuracy of the LO models is unknowable. Future collections of occurrence data should include the locations of true negatives – i.e., locations were a species is known to be absent. Furthermore, our indices could be improved by collecting data on particular key species or species' life stages. We lack, for example, occurrence data for native Olympia oyster; our oyster occurrence data are for nonnative Pacific oyster. We also lack occurrence data for rearing areas of juvenile salmonids and Dungeness crab. The juvenile life stages of these species use different habitats than their adult life stages. Rearing habitats serve essential functions that our index does not currently capture for these species. Inaccurate or noncomprehensive data can result in the mischaracterization of high value shorelines as low value and low value shorelines as high value. Both errors lead to an inefficient allocation of resources for protection and restoration of shorelines. Hennessey et al. (2011) also recommended collecting data on fisheries; habitats, marine fish, and threatened and endangered species, including state sensitive species and state species of concern.

Second, a process of validating the index should be explored. Formal model validation entails testing the accuracy of model predictions. Statistically rigorous model validation is purely objective. However, we cannot do a rigorous validation of our current index because notions of conservation "value" are normative and the current index is based on best professional judgment. Future attempts to assess the relative habitat value of shorelines could objectify value by monetizing the ecosystem services provided by the habitat functions of marine shorelines.

Third, a marine shoreline assessment that integrates structure, process, and function should be developed. Our index of relative habitat value is based on habitat functions. Habitat functions are dependent upon properties of ecosystem structures and processes, and hence, the relationships between functions and processes or structures are sometimes obscure. A quantitative model built on these relationships would provide a fuller understanding of why a place is important and insights about how to manage that place. Until we have such a model, we will improvise an integration of this assessment, which emphasizes function, with PSNERP's assessment (Cereghino et al. 2012) which emphasizes processes.

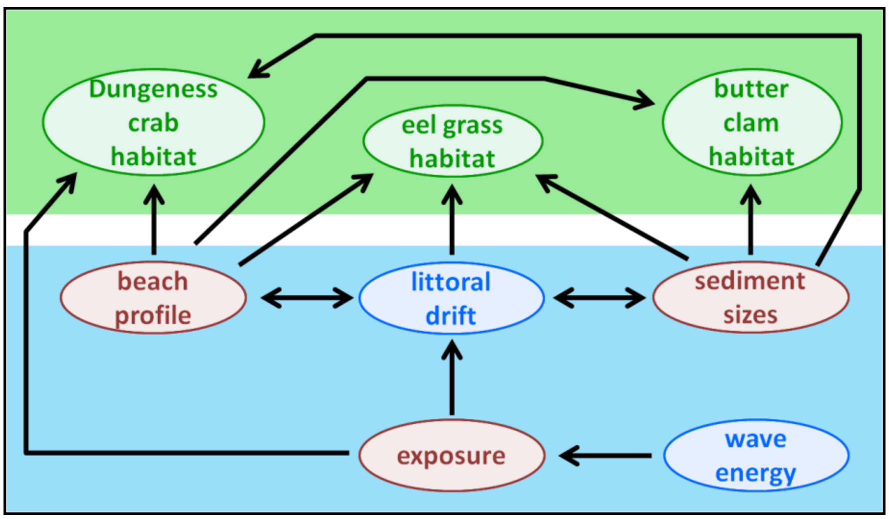


Figure 4.28. Integration of ecosystem process, structure, and function for planning and decision making. PSNERP's assessment (Cereghino et al. 2012) covers process and structure (blue area) and this assessment covers habitat functions (green area). Both assessments contribute information for understanding local shoreline ecosystems.

4.4.4. Integrating Process and Function

An assessment based solely on the occurrence on habitat functions will not provide a complete understanding of each shoreline's relative value and will neglect essential information for guiding management actions. We must also consider the ecosystem processes and structures that are responsible for creating and maintaining habitat functions (Figure 4.28). PSNERP (Cereghino et al. 2012) provides information on the condition of nearshore ecosystem processes within individual drift cells. Specifically, PSNERP assessed the relative degradation of littoral drift. In PSNERP's assessment, degradation reflects the relative loss of historical ecosystem services as indicated by landform change and shoreline modification (Table 4.6). Particular attention was given to indicators of degradation thought to be important in process dynamics.

Table 4.6. Metrics used to assess degradation of beach and barrier embayments in the PSNERP assessment (Cereghino et al. 2012).

Degradation Metrics	Degradation Index: Beach	Degradation Index: Embay-ment	Description
Lost Embayment Length		■	Loss of length was calculated as the total length of current embayment landform subtracted from the total length of historical embayment landforms within a site. While some change in length was attributed to mapping error, this metric provided a measure of gross physical change in the system to complement presence of linear stressors and nearshore zone development in barrier embayment sites.
Nearshore Impervious	■	■	The percentage of land area within 200 m of the shoreline with impervious surfaces estimated as greater than 10% was used to describe the intensity of development at a site. Development indicated by impervious surface was assumed to indicate the combination of intensive use, chronic pollution, modified hydrology, and loss of native vegetation.
Parcel Density	■		The mean number of parcels per 100m in a shoreline process unit was used to characterize both challenges and costs of negotiating protection or restoration of sediment supply and transport under complex parcel ownership, as well as chronic impacts from high density residence on vegetation and drift wood.
Sediment Supply Degradation	■	■	The sediment input degradation metric was developed by Schlenger et al. (2011) to predict the effect of overlapping stressors on the degradation of sediment input. In shoreline process units, this metric calculated the percentage of bluff-backed beach landforms located in a drift cell component showing either divergence or transport (i.e. DZ, LtR or RtL) that was covered by either fill, armoring, railroads, roads or an artificial landform, all of which PSNERP anticipated to potentially affect sediment supply budgets.
Tidal Flow Degradation		■	The tidal flow degradation metric was developed by Schlenger et al. (2011) to predict the effect of overlapping stressors on the degradation of tidal flow in embayments and river deltas. Within shoreline process units, tidal flow degradation was estimated as the percent of embayment landform length with either tidal barrier, fill, railroad, or an artificial landform.

PSNERP used their degradation assessment to develop management strategies. There were separate PSNERP strategies for deltas, coastal inlets, beaches, and barrier embayments. PSNERP mapped 16 major river deltas in Puget Sound. Because the ecological functions (existing, historical, and potentially restorable functions) of river deltas and their estuaries are universally recognized as essential and irreplaceable, deltas have been a focus of considerable attention in Puget Sound, and will continue to be a focus for the foreseeable future. Our assessment has little to contribute to the many comprehensive and detailed plans for the restoration and management of river deltas (e.g., Beamer et al. 2005, USFWS 2005, WDFW 2008b, NOAA 2009).

There are 266 coastal inlets. Human activities at coastal inlets affect their habitat functions, but activities within the inlet's entire drainage area may also have a significant impact on habitat functions. Therefore, effective management of coastal inlets must also consider land use activities far removed from the inlet's location. This will require integration of this assessment, PSNERP's assessment, and the freshwater habitats, terrestrial habitats, and water flow assessments that are part of the Puget Sound Partnership's Watershed Characterization Project. That complex process is beyond the scope of this report. Volume 3 of the Watershed Characterization Project will provide guidance for integration of multiple assessments.

The connections between shoreline habitat functions and drift cell processes are probably strongest along beaches and barrier embayments. There are 812 drift cells in Puget Sound, so some system is needed to simplify and thereby facilitate the integration of the PSNERP strategies with the results of our assessment. *Protect* is recommended for the least degraded drift cells, *enhance* for the most degraded drift cells, and *restore* for those drift cells in between (Table 4.7). To simplify the PSNERP strategies we combined the beach and barrier embayment strategies by comparing the two recommendations for each drift cell and taking the recommendation with the minimum level of degradation. To facilitate integration of the PSNERP strategies with our assessment, we calculated the mean composite index for each drift cell and then divided the drift cells into three groups by terciles. The three groups correspond to three levels of mean relative habitat value – high, medium, and low (Figure 4.29).

The integration scheme uses PSNERP's three management recommendations and the three levels of relative habitat value. The combination of the PNSERP strategy and habitat value should help to further refine management priorities within sub-basins. Where relative habitat value and the level of degradation are congruent (i.e., highest habitat & lowest degradation, intermediate habitat & intermediate degradation, lowest habitat & highest degradation), which occurs for about one-third of drift cells, then site-level management decisions should be straightforward. That is, a drift cell recommended for protection with high relative habitat value should be a higher priority for protection than drift cell recommended for protection with a medium or low relative habitat value. Likewise, a drift cell recommended for restoration with high habitat value should be a higher priority for restoration than drift cell recommended for restoration with a medium or low value.

When the assessments are not congruent, what is the management recommendation? For instance, how should we manage shorelines that have are highly degraded shoreline processes (enhance recommendation) but high relative habitat value (which describes about 11 percent of drift cells)? "Enhance" signifies a low priority for protection or restoration but high habitat conservation value contraindicates that recommendation. Site-level management decisions for these drift cells will require further analysis and a reappraisal of conservation objectives. Local information is important for all site-level decisions, but will be especially important under these circumstances.

Why would the assessments be incongruent? For instance, why would a drift cell have high degradation and high habitat value or low degradation and low habitat conservation value? There are several potential explanations. First, there may be time lags in the responses of fish, wildlife, and plant species to the degradation of nearshore processes. In other words, it may take some time for a degraded drift cell to lose its habitat functions. Second, certain fish and wildlife species may be responding to ecosystem structures and processes other than those related to littoral drift. For instance, they may be responding to the proximity of nearby rocky reefs, upwelling currents, or local fetch. Third, species could be responding to structures or processes that occur at spatial scales different than those of our assessment. And finally, the two assessments may be incongruent because one or both of them are wrong. Even the best models are occasionally wrong, and hence, there will be portions of the Puget Sound shoreline (hopefully very small portions) where the assessments are wrong.

Table 4.7. Relationship between drift cell degradation and management recommendations from the Puget Sound Nearshore Ecosystem Restoration Project (PSNERP; Cereghino et al. 2012).

Low Degradation	Moderate Degradation	High Degradation
Protection	**Restoration**	**Enhancement**
Site likely provides substantial ecosystem services in its existing state. The primary goal of management is to prevent and substantial loss of ecosystem processes or functions.	Site where indicators of degradation suggest the opportunity to substantively increase ecosystem services through restoration.	Site where the level of degradation appears to be so intense that restoration of self-sustaining and resilient ecosystem services may be severely compromised. Focus on enhancement of critical habitat functions.

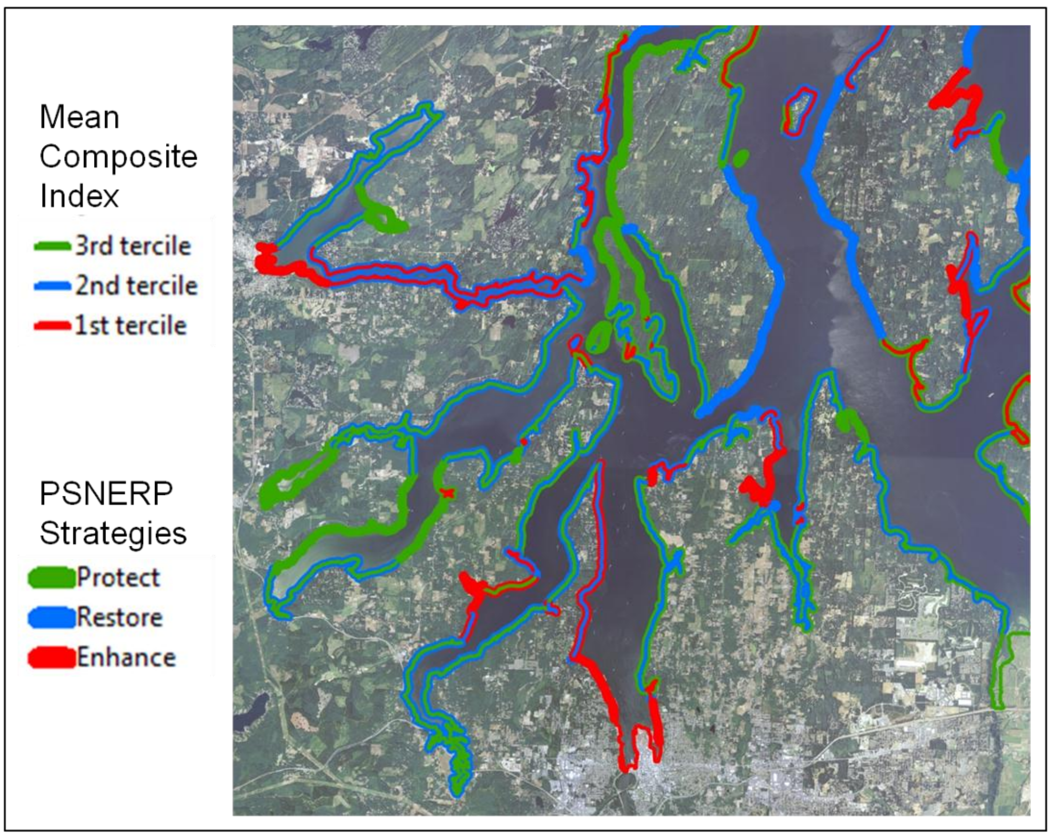


Figure 4.29. Integration of information from the marine shoreline assessment and PSNERP's assessment (Cereghino et al. 2012). Overlay of mean composite index onto the PSNERP strategies for beaches and embayments. The composite index was averaged over each drift cell. Terciles for mean composite index (thinner center line) were calculated for each sub-basin. Green is highest relative habitat value and red is lowest relative value. Beach and embayment strategies (thicker line) were combined by comparing strategies for each drift cell and taking the recommendation with the minimum level of degradation: protect < restore < enhance. Management recommendations for deltas (Nisqually and Deschutes rivers) not shown.

Part 5: Summary and Assessment Utilization

5.1. Summary

We conducted a coarse-scale assessment of relative value of places for the conservation of terrestrial, freshwater, and marine shoreline habitats in the Puget Sound Basin. The assessment was broken into three separate assessments because these three environments exhibit significant differences in spatial dimensions, spatial scale, data quality, and ecosystem-level processes. We collected no new empirical data for the assessments. We selected existing data that met our needs, analyzed it to answer a specific question, and generated new information that should be useful to local governments in answering that question. The simplest expression of that question is: In Puget Sound Basin, where should new future development not be located? Overall, the answers were unsurprising, but nevertheless provide scientifically-based information to guide future land use planning.

The main result of our coarse-scale terrestrial assessment was that relative conservation value exhibits an obvious spatial gradient. Relative conservation value has its minimum in the Seattle-Tacoma metropolis and is at its maximum in the large blocks of undeveloped forest land that begin in the foothills of the Cascades and Olympic Mountains. In other words, at the landscape scale, relative conservation value follows the classic urban-to-wildland gradient (Marzluff et al. 2001). In Pierce, King, and Snohomish counties that gradient runs roughly west-to-east. Major exceptions to this gradient pattern are: 1) mouths of major rivers that are relatively undeveloped, such as the Nisqually, Skagit, and Nooksack, which support large concentrations of waterfowl and shorebirds; and 2) the oak-grassland habitat types in and around Fort Lewis.

The results of the terrestrial assessment highlight a rural or exurban "decision space" sandwiched between the densely-populated urban areas on Puget Sound and the relatively undeveloped foothills. These rural areas contain open spaces, both agricultural and timberland, that provide habitats for native fish and wildlife species. Over the coming decades local governments will decide where and what types of development may occur within that decision space. Our assessments can inform the decisions regarding where.

The freshwater assessment's index of relative conservation value was comprised of three components: the density of hydrogeomorphic features, local salmonid habitats, and accumulative downstream habitats. High values for two of the three components were situated predominately in non-overlapping portions of each WRIA: AUs in valley bottoms had the highest density of hydrogeomorphic features, and AUs in foothills and mountains had the biggest impact on accumulative downstream habitats. Because two of the components dominated in different parts of a WRIA and the third component covered many other parts of each WRIA, the assessment showed that nearly 75% of AUs in most WRIAs obtained at least a moderate score for at least one component. Therefore, according to our assessment, residential development that aims to conserve freshwater habitats will be very challenging because nearly all AUs in a WRIA contribute in some way to the quality of freshwater habitats. On the other hand, because the relative conservation value of AUs spans a wide range, the results of the freshwater assessment can be used to direct future development away from AUs that are the most valuable for the conservation of freshwater habitats.

The results of the freshwater assessment are, in part, a consequence of our model which incorporated stream network connectivity. Stream network connectivity and the ubiquity of salmonids pose tremendous challenges for future land use planning that strives to conserve freshwater habitats while accommodating human population growth. To effectively address these challenges, holistic approaches to watershed management (Healy 1998) should be incorporated into local government comprehensive plans.

Like the freshwater assessment, the marine shoreline assessment demonstrated the ubiquity of places that have some value for the conservation of fish and wildlife habitats. The marine shoreline assessment's index of relative conservation value was comprised of 41 components. While very few shorelines scored high for a large proportion of components, over half of all shorelines scored moderate to high for multiple components, and over 99% of shorelines contained or were in close proximity to at least one component. This has serious implications under Washington's Shoreline Management Act (RCW 90.58). The governing principles (WAC 173-26-186) of the shoreline guidelines (WAC 173-26-176) established under the SMA state, "Local [shoreline] master programs shall include policies and regulations designed to achieve no net loss of those ecological functions." Any shoreline segment with composite index score greater than zero contains or is in close proximity to at least one ecological function. Local jurisdictions must address the protection of habitat functions, and as the data show, habitat functions occur nearly everywhere along the shoreline of Puget Sound. However, the type and degree of protection required for each habitat function will vary greatly.

While the marine shoreline assessment shows that nearly all marine shorelines contains or are in close proximity to at least one habitat function, the assessment does not account for all habitat functions. For instance, we lacked occurrence data for the nearshore rearing habitats of juvenile salmonids and the rearing habitats of juvenile Dungeness crab. These are essential habitat functions that support vital commercial and recreational fisheries. Investments in data collection are needed to map the locations and quality of these and other habitat functions.

The coarse-scale spatial patterns of relative conservation value resulting from the separate habitat assessments were quite different, and these incongruities may pose substantial challenges for land use planning. For example, according to the freshwater assessment, the most valuable places for the conservation of salmonids are located in valley bottoms, in particular, wide undeveloped[18] valley bottoms. In contrast, the terrestrial assessment indicated that the most valuable places were generally located in the foothills of the Cascades and Olympic Mountains. Hence, it seems local governments may need to more carefully evaluate ecological trade-offs when deciding where and how to accommodate future human population growth.

Integration of Assessments

The Department of Ecology has completed two assessments for water resources – water flow and water quality; and WDFW has completed three assessments for habitats – terrestrial, freshwater, and marine shorelines. Used separately, these assessments provide valuable information for land use planning, however, effective land use planning ultimately demands integrated information that provides a comprehensive description of the landscape or watershed. Assessments for water resources and habitats were done separately by Ecology and WDFW, but our efforts have been coordinated. Both agencies analyzed the same region using the same assessment units, and where appropriate, used the same spatial data (e.g., wetlands). This coordination facilitates integration but does not completely solve the integration problem. Integration of the five assessments and of our assessments with other assessments (e.g., Cereghino et al. 2012) will be covered in volume 3 of the Puget Sound Watershed Characterization Project.

[18] Recall that in our freshwater assessment "undeveloped" included agricultural and managed timberlands.

Integrating several assessments should lead to more robust land use planning recommendations. If, for instance, multiple assessments all indicate that a place (i.e., AU) is relatively high value, then planners can be more certain that that place is indeed valuable and can be more confident about the appropriate land use designation.

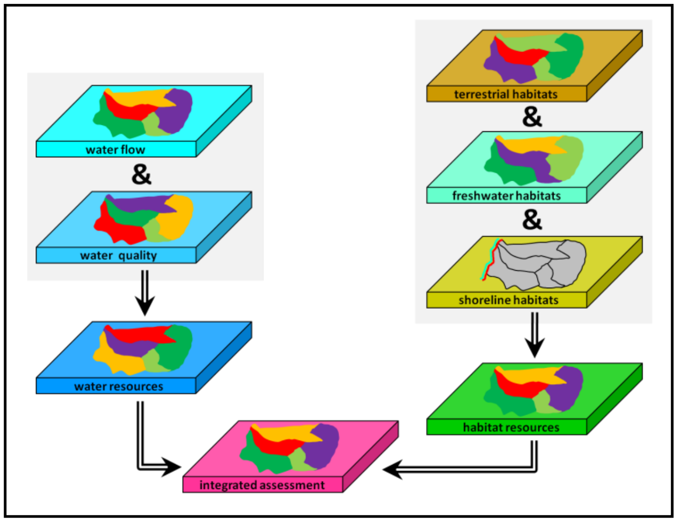


Figure 5.1. An idealized integration of assessments. Water flow and water quality assessments are integrated into an overall "water resources" assessment. Terrestrial, freshwater, and marine shoreline habitats assessments are integrated into an overall habitat resources assessment. However, such integrated assessments must always be decomposed into their components in order to more fully understand the resource conditions and values in each AU.

Integrating the water resources and habitats assessments could consist of simple map overlays or a quantitative combination of indices that yields an overall integrated index (Figure 5.1). Integration can indicate where high value places for both water resources and habitats coincide or where low value places for both water resources and habitats coincide. However, integration can also obscure important information such as when a place is high value for one assessment but low value for the other assessments. For this reason, interpretation of an integrated assessment must be supplemented with information from each of the individual assessments.

AUs with the same relative conservation value may not be valuable for the same reasons. AUs that are highly important for water flow and other AUs having high relative conservation value for terrestrial habitats could both be managed to protect their valuable natural resources but managed in different ways. The geodatabases associated with the assessments can be queried to determine why an AU was assigned its relative value. Land use plans for AUs should not be developed until reasons for each AU's relative value are known and understood. The appropriate land uses and management within an AU should be determined by the relative value of water resources and habitats within the AU.

5.2 Using the Assessments

The Puget Sound Watershed Characterization encompasses a set of spatially-explicit assessments that combine multiple data sources covering the entire Puget Sound Basin. These assessments are of necessity coarse-scale because they cover a very large region and generalized because they utilize data collected by remote sensing (e.g., satellite) or by field surveys for a purpose originally not connected with the Puget Sound Watershed Characterization. Nonetheless, the chosen data sets and the manner in which they were combined provide a potentially useful, regional-scale perspective on the spatial distribution of relative conservation value that is not generally provided by other available tools.

The main application of these assessments is to guide local land use zoning that occurs at the scale of 100s to 1000s of acres. County governments should use the results of the terrestrial and freshwater assessments to direct expansion of urban growth areas or new residential development away from places with relatively high conservation value for fish and wildlife habitats. Conversely, the first places to develop or develop more densely are those areas (i.e., AUs) at the low end of relative conservation value. New development should be avoided in areas at the highest end of conservation value. When directing new development toward lower value areas, local jurisdictions must still institute policies and regulations that protect the functions and values of critical areas.

The results of the marine shoreline habitats assessment can be used to guide the designation of shoreline land use zones that will achieve no net loss of habitat functions that currently exist along shorelines. The marine shoreline assessment can also help prioritize shoreline restoration within oceanographic sub-basins.

The main application of the assessments is local land use planning by county and city governments, however, the assessments can inform a wide range of local conservation efforts. Potential applications include zoning for transfer of development rights programs or targeting specific areas for conservation easements. For local governments planning under the Growth Management act, the assessments can be used as "best available science" (as described under WAC 365-195-905) when they update their comprehensive plans. Because the individual assessment units are typically more than 5 square miles in area, where to focus habitat protection or residential development cannot be delineated more precisely without additional information. Similarly, guidance on "what to do" will require additional local information that is not included in the assessments.

We must emphasize that the assessments do not provide the solution to complicated policy issues. The assessments simply provide information that can inform public processes intended to resolve policy issues, such as issues related to local land use planning. Decisions regarding land use policy, for instance, must simultaneously weigh multiple economic, social, and ecological factors. Deciding where to locate new residential development often entails trade-offs among these factors. Science has an important role to play, but ultimately such decisions are based on societal values (Wilhere 2008). Scientific information can lead to smarter decisions but only when the information is used properly. To help local governments make the best use of our assessments, Ecology and WDFW have established the Watershed Characterization Technical Assistance Team comprised of scientists with various expertise in the natural sciences. The team is available to assist local governments in using the assessments and integrating them with other locally available information or data.

5.2.1 Interpreting the Results

This assessment provides a county-scale or WRIA-scale perspective on the *relative* value of small watersheds (i.e., AUs) or shoreline segments for the conservation of terrestrial, freshwater, or marine shoreline habitats. The primary products of the assessment are maps that show the relative value of the AUs or shorelines. Their relative values are expressed through quantitative indices that can be used to rank AUs within a county, WRIA, or oceanographic sub-basin.

Spatial Extent of the Assessments

Although we assessed relative conservation value for the entire Puget Sound Basin, the spatial extents of each assessment were different. The spatial extent over which each assessment was conducted affects one's interpretation of relative conservation value. For instance, an AU could have moderate relative value in the Puget Trough Ecoregion but have the highest relative value in Pierce County. A land use plan for Pierce County might target that AU for protection, but a conservation plan for the Puget Trough Ecoregion might not. An AU with high regional value should be considered more valuable to regional authorities than an AU that has high local value but only low or moderate regional value. On the other hand, an AU with low regional value could be the most valuable AU within a local jurisdiction. Local authorities should keep that in mind when using this assessment and identify the most valuable AUs within their planning areas.

Wildlife do not recognize geopolitical boundaries and most are unimpeded by watershed boundaries. Hence, the extent of our terrestrial assessment covered the entire Puget Sound Basin with no spatial sub-divisions. Hence, results from the terrestrial assessment allow valid comparisons among AUs in different WRIAs or different counties.

For the freshwater habitats assessment we divided the Puget Sound Basin along WRIA boundaries, and relative conservation value index was relative within each WRIA. Most WRIAs support unique salmonid stocks and we wanted to maintain that separation. There are 19 WRIAs in Puget Sound Basin, but we lumped the lower Skagit and upper Skagit WRIAS (WRIAs 3 and 4) into a single WRIA. Our freshwater assessment in its current form does not enable Puget Sound Basin-wide comparisons.

For the marine shoreline habitats assessment, we split the Sound into seven oceanographic sub-basins, and relative conservation value index was relative within each sub-basin. Hence, scores among shoreline segments in different sub-basins are not comparable, and our assessment in its current form does not enable Puget Sound wide comparisons. Regional authorities should keep that in mind when using this assessment to identify the most valuable shorelines within Puget Sound. The assessment could be restructured and reformulated to provide Sound-wide comparisons.

Mapping Relative Conservation Value

The index of relative conservation value is a continuous value from 0 to 1 (or 0 to 100). For the purposes of mapping, the continuous values can be divided into categories via equal intervals or statistical quantiles (Figure 5.2). Interval widths or quantile sizes are somewhat arbitrary, and therefore, one must be cautious when interpreting maps. For instance, an AU with a score of 0.78 would be in the "highest value" category when using 4 equal intervals but be in a lesser category when using 5 equal intervals. Furthermore, categories obscure some quantitative relationships between AUs. When using 5 equal intervals, for instance, an AU with a score of 0.61 is in the same category as an AU with a score of 0.79 but that AU is actually more similar to an AU with a score of 0.59 which is in a different category.

Because some particularly high-value sites can be outliers, the distribution of index values can be greatly skewed. In the marine shoreline assessment, for instance, the highest value shoreline segment in the Juan de Fuca sub-basin had a score 1 (by definition) but the next highest score was 0.64. For situations such as these, dividing the continuous values into quantiles provides a uniform categorical ranking of sites.

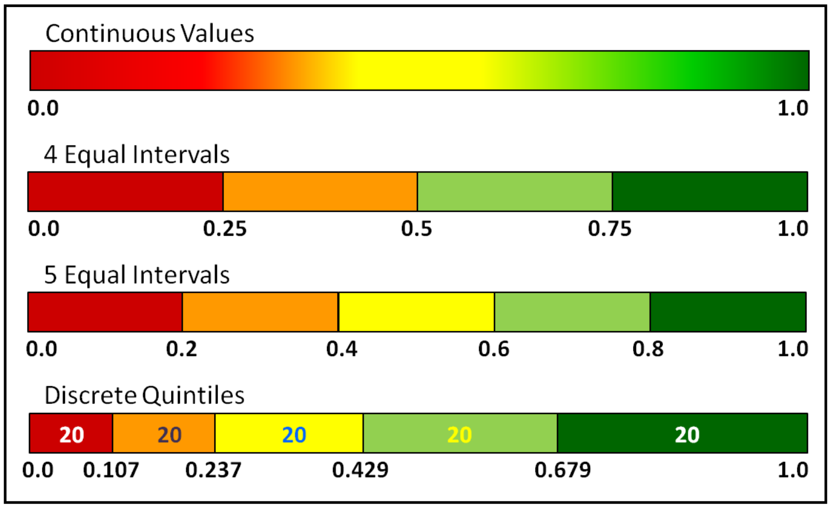


Figure 5.2. Interpretation of maps. The indices of relative conservation value are continuous variables between 0 and 1. To facilitate the interpretation of indices, the continuous variables are divided into categories via equal intervals or statistical quantiles. Quartile and quintiles, for instance, are quantiles in which each group contains 25% and 20% of assessment units, respectively. These simplifications warrant two caveats. First, within categories segments do not have the same conservation value. Second, the categories are somewhat arbitrary and can obscure relationships between segments.

Uncertainty

Uncertainty is inherent to every management decision affecting wildlife and their habitats (Wilhere 2012). Our assessments of relative conservation value are also uncertain. Greater uncertainty does not mean that an AU has lesser value than that calculated through the assessment, although it might have lesser value. Greater uncertainty only means that the actual relative conservation value could be somewhat larger or smaller than the calculated value.

We can make generalizations about when we can be more confident about scores and when we should be more wary. For assessments of this type, relative values at the extreme ends of their range (e.g., the top and bottom deciles) tend to have the smallest uncertainty, and places with more moderate scores tend to have larger uncertainty. Consequently, if our assessment indicates that an AU is only slightly more (or less) valuable than another AU, then both AUs should be treated as if they have the same relative value. This is especially true when the AUs have moderate values between 0.3 and 0.7.

5.2.2 Caveats

When using the results of the terrestrial, freshwater, and marine shoreline assessments local governments should keep the following in mind.

Other Assessments

This assessment and all the assessments done for the Puget Sound Partnership's Watershed Characterization project do not constitute all the information necessary and sufficient to address natural resources conservation through land use planning. Therefore, this assessment should be supplemented with other assessments. For example, the Puget Sound Nearshore Ecosystem Restoration Project has recently completed a sound-wide assessment (Cereghino et al. 2012) which complements our marine shorelines assessment. Furthermore, local governments may wish to fund their own finer-scale assessments. May and Peterson (2003) for salmonid habitats in Kitsap County and Diefenderfer et al. (2009) for marine shorelines in Jefferson County are two examples of local governments funding high-quality assessments.

Each of the Salmon Recovery Lead Entities has done their own assessments to support their recovery plans for Chinook and steelhead (e.g., East Kitsap 2004, Pierce County 2004, Snohomish County 2005). The work done by lead entities serves a particular purpose, is highly attuned to local knowledge, and has involved local stakeholders, and therefore, our assessment is not a substitute for the assessments and plans of the lead entities.

Assembling, organizing, and integrating scientific information from diverse sources is a continual challenge for local governments. In volume 3 of the watershed characterization project, we will describe methods to effect this integration. In addition, to assist local governments overcome this challenge, WDFW and Ecology have formed a watershed characterization technical assistance team (WCTAT) consisting of state agency scientists with expertise in wildlife biology, fish biology, wetlands, hydrology, geomorphology, and modeling.

Limitations of the Assessments

The terrestrial, freshwater, and marine shoreline assessments have the following major limitations. First, our assessments are landscape-scale assessments, and consequently, do not address habitat issues that are best addressed through finer-scale studies. Finer-scale or site-level actions, such as critical area ordinances that protect nest sites, riparian areas, or wetlands, will remain essential to the success of local habitat conservation efforts. We did our assessments with the expectation that finer-scale studies will be done by county governments as the need arises. When developing land use plans, city and county governments should evaluate the need for finer-scale assessments and conduct them where needed.

Second, our indices of relative conservation value are not comprehensive. The assessments are not comprehensive in three respects. First, the assessments did not explicitly include all species because we lack reasonably accurate occurrence data for most species, even for most priority species such as Keen's myotis, pileated woodpecker, band-tailed pigeon, western toad, and Pacific lamprey. Second, the assessments did not fully address habitat connectivity because it has been or will be addressed through other assessments, as explained below. Third, we narrowed the spatial extent of the terrestrial and freshwater assessments to areas that fall under the jurisdiction of city and county land use plans. That is, we did not address species or habitats that are mostly confined to higher elevations (>2000 ft) on public lands.

In the terrestrial and freshwater assessments, however, we did implicitly address nearly all species. In the terrestrial assessment, relative conservation value was mainly a function of landscape integrity. The presence of PHS habitats was also an important factor but because of their relatively small spatial extent, PHS habitats were much less influential than landscape integrity. Basing conservation value on landscape integrity was essentially a coarse-filter approach which assumes that areas with higher landscape integrity will provide higher quality habitat for the majority of wildlife species. In the freshwater assessment, conservation value was based largely on the quantity and quality of salmonid habitats. This was effectively an umbrella-species approach which assumes that areas protected for salmonid habitats will also protect habitats for the majority of other species in lotic habitats.

In the marine shoreline assessment, conservation value was calculated as a composite index consisting of 41 diverse components. This was effectively a richness approach which assumes that our 41 components can serve as adequate surrogates for the majority of species in marine shoreline habitats of Puget Sound. However, the available occurrence data were biased towards harvested species. Therefore, the marine assessment might be better characterized as an ecosystem services approach in which habitats are ecosystem functions that provide the ecosystem service of food provision.

Because of the assumptions and simplifications we made, the terrestrial, freshwater, and marine shoreline assessments may not adequately address the particular habitat needs of rare or imperiled (i.e., state or federally listed) species or species highly susceptible to human disturbance. If rare or imperiled species inhabit a local jurisdiction, then the special needs of such species should be specifically addressed in local land use plans.

Third, one particularly important aspect of biodiversity conservation which we did not adequately address was connectivity. Landscape integrity in the terrestrial assessment incorporated factors which address habitat connectivity but only obliquely. More detailed assessments on habitat connectivity may be necessary. Connections between habitat patches can be provided by smaller-scale features such as riparian corridors that are best delineated through finer-scale assessments. State-wide connectivity has been addressed by the Washington Wildlife Habitat Connectivity Working Group (WHCWG 2010). The results of the WHCWG assessment should be incorporated into regional land use planning. Upstream and downstream relationships within the stream network were foundational to the freshwater assessment, however, that assessment did not explicitly include artificial barriers to fish passage. The freshwater assessment addressed stream connectivity indirectly through occurrence data that documented the presence of andromous salmonids and expert judgments about which streams could support anadromous fish when artificial barriers are removed.

Along marine shorelines the main connectivity issue is the movement of sediments within littoral drift cells. Maintaining process connectivity within drift cells has been the focus of PSNERP. Integrating our marine shoreline assessment with PSNERPs assessments of drift cells will help local governments prioritize shorelines for protection and restoration of connectivity.

Fourth, our assessment is essentially an approximate snapshot of current conditions. We did not project changes in land use or habitats into the future, nor did we estimate the risk of adverse changes to habitat due to climate change. Where available, assessments of future potential changes due to climate change (e.g., Lee and Hamlet 2011) should be integrated into local land use planning.

Fifth, there is no purely objective “conservation value” that can be empirically validated. “Value” is based on one’s belief about what is valuable, and therefore, subjective. Furthermore, there is a wide variety of potential credible models of conservation value that could be constructed for this assessment. Our models of habitat conservation value for the three assessments were based on a number of subjective judgments for which there was uncertainty: which factors to include, their relative influence, and how to assemble them. Through numerous meetings with experts and intensive peer review we believe we have developed scientifically credible indices of relative conservation value. By “scientifically credible” we mean that we used the best available data, developed models based on well-established concepts, combined the data through standard statistical procedures, verified the results, and subjected all our work to a peer-review process. Nevertheless, conservation value is ultimately a normative concept and future assessments should seek guidance from policy makers.

Lastly, as data, technology, and knowledge improves over time better assessments will emerge. Other initiatives, separate from those of the Puget Sound Partnership, will reassess habitats in the Puget Sound Basin. For Instance, the Western Governors’ Association has initiated a project to identify “crucial habitats” throughout the western states (WGWC 2011). WDFW is participating in that initiative, and the results of the crucial habitats assessment could supplement or supplant this assessment.

5.3. Improving the Assessments

The major limitations of our terrestrial, freshwater, and marine shoreline assessments are mainly due to: 1) lack of adequate occurrence data for the vast majority of vertebrate fish and wildlife species in the Puget Sound Basin, 2) lack of models for estimating the spatial distributions of fish and wildlife species and their various life stages, and 3) lack of guidance from policy makers regarding the meaning of relative conservation value.

Empirical data on the locations of wildlife species collected by WDFW and other agencies generally focus on imperiled (i.e., federal or state listed) species or harvested species. However, "biodiversity" encompasses hundreds more animal species than those species for which we regularly conduct comprehensive, systematic surveys. With the notable exception of the annual PSAMP bird surveys (Nysewander et al. 2005), there are no systematic surveys of vertebrate species covering the entire Puget Sound Basin. Even most priority species listed under WDFW's Priority habitats and Species Program (WDFW 2008a) lack adequate data on their occurrences in Washington. The presence (or absence) of priority species should be a major factor in determining the relative conservation value of places. Investing in comprehensive, systematic surveys for priority species, especially species considered to be umbrella or indicator species, would greatly improve regional and local assessments of relative conservation value.

Collecting accurate data on species' occurrences is expensive, and consequently, the shortcomings of our data are common to fish and wildlife datasets everywhere. A practical alternative to comprehensive, systematic surveys is modeling. Developing models that predict either species occurrences or their habitats is much less expensive than comprehensive surveys, but there is a trade-off – maps based on models have greater uncertainty than maps based entirely on survey data. Species occurrence models are based on occurrence data, but the survey effort needed is much less than that associated with comprehensive surveys. However, a model's uncertainty can be greatly reduced by collecting more occurrence data. Investing in a relatively small survey effort to develop occurrence models for key vertebrate species would greatly improve regional and local assessments of relative conservation value.

We believe we have developed scientifically credible indices of relative conservation value, however, conservation "value" is based on beliefs about what is valuable, and therefore, is ultimately normative. Imperiled species such as the northern spotted owl, harvested species such as coho salmon, and common species such as acorn barnacle are all valuable in some way, however, society has made choices that treat each of these species and their habitats differently. Likewise, when assessing the relative conservation value of places we may wish to treat species occurrences or their habitats differently. The relative value assigned to the occurrences or habitats of various species is a policy decision, and should be made by policy makers. Our current assessments avoided such decisions by treating all species or habitats equally (i.e., equal weights of importance), but future assessments might be improved by working with policy makers to assign importance weights to species or habitat based on societal values. Importance weights could be based on the monetary value of ecosystem services provided by each species or habitat. That approach would require an investment in the modeling and estimation of ecosystem services.

References

Agee, J.K. 1993. Fire Ecology of Pacific Northwest Forests. Island Press, Washington, DC.

Alan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35:257-284.

Alexander, D., and R. Tomalty. 2002. Smart growth and sustainable development: challenges, solutions and policy directions. Local Environment 7:397-409.

Andreasen, J.K., R.V. O'Neill, R. Noss, N.C. Slosser. 2001. Considerations for the development of a terrestrial index of ecological integrity. Ecological Indicators 1:21-35

Audubon. 2001. Important Bird Areas of Washington. compiled by T. Cullinan. Audubon Washington, Olympia, Washington.

Azerrad, J. J. Carleton, J. Davis, T. Quinn, C. Sato, M. Tirhi, S. Tomassi, and G. Wilhere. 2009. Landscape Planning for Washington's Wildlife: Managing for Biodiversity in Developing Areas. Washington Dept. of Fish and Wildlife, Olympia, Washington.

Bartz, K.K., K.M. Lagueux, M.D. Scheuerell, T. Beechie, A.D. Haas, and M.H. Ruckelshaus. 2006. Translating restoration scenarios into habitat conditions: an initial step in evaluating recovery strategies for Chinook salmon (*Oncorhynchus tshawytshca*). Canadian Journal of Fisheries and Aquatic Sciences 63:1578-1595.

Beamer, E., A. McBride, C. Greene, R. Henderson, G. Hood, K. Wolf, K. Larsen, C. Rice, and K. Fresh. 2005. Delta and nearshore restoration for the recovery of wild Skagit River Chinook salmon: linking estuary restoration to wild Chinook salmon populations. Skagit River System Cooperative, LaConner, Washington.

Bender, D.J., L. Tischendorf, and L. Fahrig. 2003. Using patch isolation metrics to predict animal movement in binary landscapes. Landscape Ecology 18:17-39.

Berry, H.D., J.R. Harper, T.F. Mumford, B.E. Bookheim, A.T. Sewell, L.J. Tamayo. 2001a. The Washington State ShoreZone Inventory User's Manual. Nearshore Habitat Program, Washington State Department of Natural Resources, Olympia, Washington.

Berry, H.D., J.R. Harper, T.F. Mumford, B.E. Bookheim, A.T. Sewell, and L.J. Tamayo. 2001b. Washington Shorezone Inventory Data Dictionary. Nearshore Habitat Program, Washington State Department of Natural Resources, Olympia, Washington.

Box, G.E.P., and N.R. Draper. 1987. Empirical Model-Building and Response Surfaces. John Wiley & Sons, New York, New York.

Brown, M.T., and M.B. Vivas. 2005. Landscape development intensity index. Environmental Monitoring and Assessment 101:289-309.

Burnett, K.M., G.H. Reeves, S.E. Clarke, and K.R. Christiansen. 2006. Comparing riparian and catchment influences on stream habitat in a forested, montane landscape. American Fisheries Society Symposium 48:175-197.

Burnett, K.M., G.H. Reeves, D.J. Miller, S. Clarke, K. Vance-Borland, and K. Christiansen. 2007. Distributions of salmon-habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications 17:66-80.

Busch, D.S., M. Sheer, K. Burnett, P. McElhany, and T. Cooney. 2011. Landscape-level model to predict spawning habitat for lower Columbia River fall Chinook salmon (*Oncorhynchus tshawytscha*). River Research and Applications *in pre*ss.

Cassidy, K. M. 1997. Land cover of Washington State: Description and Management. Volume 1 in Washington State Gap Analysis - Final Report. Washington Cooperative Fish and Wildlife Research Unit, University of Washington, Seattle.

C-CAP. 2008. C-CAP zone 1 2006-era land cover metadata. Coastal Change Analysis Program, NOAA Coastal Services Center, Charleston, South Carolina. accessed at http://csc.noaa.gov/dataviewer/webfiles/metadata/z1_2006.html

Cereghino, P., J. Toft, C. Simensted, E. Iverson, S. Campbell, C. Behrens, J. Burke, and B. Craig. 2012. Strategies for Nearshore Protection and Restoration in Puget Sound. Technical Report No. 2012-01. U.S. Army Corps of Engineers, Seattle District and Washington State Department of Fish and Wildlife, Olympia, Washington.

Clarke, S.E, K.M. Burnett, and D.J. Miller. 2008. Modeling streams and hydrogeomorphic attributes in Oregon from digital and field data. Journal of the American Water Resources Association 44:459-477.

Cooney, T., and D. Holzer. 2006. Interior Columbia Basin stream type Chinook salmon and steelhead populations: habitat intrinsic potential analysis. Appendix C in Viability Criteria for Application to Interior Columbia Basin Salmonid ESUs, Interior Columbia Basin Technical Recovery Team, Northwest Fisheries Science Center, NOAA Fisheries, Portland, Oregon.

Corps of Engineers. 2005. Levee centerlines within the civil works boundaries of the U.S. Army Corps of Engineers. U.S. Army Corps of Engineers, Seattle, Washington.

Cowardin, L.M., V. Carter, F.C. Golet, and E.T. LaRoe. 1979. Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31. U.S. Fish and Wildlife Service, Washington, D.C.

Daniels, T. 2001. Smart growth: a new American approach to regional planning. Planning Practice & Research 16:27-279.

Dethier, M.N. 1990. A Marine and Estuarine Habitat Classification System for Washington State. Natural Heritage Program, Washington Department of Natural Resources, Olympia, Washington.

Dethier, M. 2006. Native Shellfish in Nearshore Ecosystems of Puget Sound. Puget Sound Nearshore Partnership Technical Report No. 2006-04. U.S. Army Corps of Engineers, Seattle, Washington.

DeGasperi, C.L., H.B. Berge, K.R. Whiting, J.J. Burkey, J.L. Cassin, and R.R. Fuerstenberg. 2009. Linking hydrological alteration to biological impairment in urbanizing streams of the Puget Sound lowland Washington, USA. Journal of the American Water Resources Association 45:512-533.

Diamond, JM. 1975. The island dilemma: lessons of modern biogeographic studies for the design of natural reserves. Biological Conservation 7:129-146.

Diefenderfer, H.L., K.L. Sobocinski, R.M. Thom, C.W. May, A.B. Borde, S.L. Southard, J. Vavrinec, and N.K. Sather. 2009. Multiscale analysis of restoration priorities for marine shoreline planning. Environmental Management 44:712-731.

DNR. 2006. Washington State Watercourse Hydrography. Forest Practices Division, Washington State Department of Natural Resources, Olympia, Washington.

DNR. 2011. Washington DNR Transportation Data Layer. Forest Practices Division, Washington State Department of Natural Resources, Olympia, Washington.

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.

East Kitsap Lead Entity. 2004. The East Kitsap Peninsula Lead Entity Salmon Recovery Strategy. Kitsap County Department of Community Development, Port Orchard, Washington.

Esselman, P.C., D.M. Infante, L. Wang, D. Wu, A.R. Cooper, and W.W. Taylor. 2011. An index of cumulative disturbance to river fish habitats of the conterminous United States from landscape anthropogenic activities. Ecological Restoration 29:133-151.

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics. 34:487-515.

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

Feist, B.E., E. A. Steel, G.R. Pess, and R.E. Bilby. 2003. The influence of scale on salmon habitat restoration priorities. Animal Conservation 6:271-282.

Firman, J.C., E.A. Steel, D.W. Jensen, K.M. Burnett, K. Christriansen, B.E. Feist, D.P. Larsen, and K. Anlauf. 2011. Landscape models of adult coho salmon density examined at four spatial scales. Transactions of the American Fisheries Society 140:440-455.

Fleishman, E., D.D., Murhpy, and P.F. Brussard. 2000. A new method for selection of umbrella species for conservation planning. Ecological Applications 10:569-579.

Floberg, J., M. Goering, G. Wilhere, and 16 other authors. 2004. Willamette Valley-Puget Trough-Georgia Basin Ecoregional Assessment. Prepared by The Nature Conservancy and Washington Department of Fish and Wildlife. The Nature Conservancy, Seattle, Washington.

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 15:212-231.

Frissell, C., P.H. Morrison, S.B. Adams, L.H. Swope, and N.P. Hitt. 2000. Conservation Priorities: an Assessment of Freshwater Habitat for Puget Sound Salmon. Prepared at the request of the Trust for Public Land, Northwest Regional Office, Seattle, Washington.

Fry, J., G. Xian, S. Jin, J. Dewitz, C. Homer, L. Yang, C. Barnes, N. Herold, and J. Wickham. 2011. Completion of the 2006 National Land Cover Database for the Conterminous United States. Photogrammetric Engineering and Remote Sensing 77:858-864.

Gergel S.E., M.G. Turner, J.R. Miller, J.M. Melack, and E.H. Stanley. 2002. Landscape indicators of human impacts to riverine systems. Aquatic Science 64:118-128.

Glennon, M.J., and W.F. Porter. 2005. Effects of land use management on biotic integrity: an investigation of bird communities. Biological Conservation 126:499-511.

Goetz, F., C. Tanner, C.S. Simenstad, K. Fresh, T. Mumford, and M. Logsdon. 2004. Guiding restoration principles. Puget Sound Nearshore Partnership Technical Report No. 2004-03. Published by Washington Sea Grant Program, University of Washington, Seattle, Washington.

Gustafson E.J., and G.R. Parker. 1994. Using an index of habitat patch proximity for landscape design. Landscape and Urban Planning 29:117-30.

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.

Healey, M.C. 1998. Paradigms, policies and prognostication about the management of watershed ecosystems. pp. 662-682 in R.J. Naiman and R. Bilby (ed.), Ecology and management of streams and rivers in the Pacific coastal ecoregion. Springer-Verlag, New York, New York.

Hennessey, J., B. Nichols, and the State Ocean Caucus. 2011. Marine Spatial Planning in Washington: Final Report and Recommendations of the State Ocean Caucus to the Washington State Legislature. Publication no. 10-06-027. Washington Department of Ecology, Olympia, Washington.

Howes, D.E., J.R. Harper, and E.H. Owens. 1994. Physical shore-zone mapping system for British Columbia. B.C. Ministry of Environment, Lands and Parks, Victoria, British Columbia.

Hughes, R., M., S. Howlin, and P.R. Kaufmann. 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. Transactions of the American Fisheries Society 133:1497-1515.

Iachetti, P., J. Floberg, G. Wilhere, and 12 other authors. 2006. A Conservation Assessment for the North Cascades and Pacific Ranges Ecoregion. Prepared by the Nature Conservancy of Canada, The

Nature Conservancy, and the Washington Department of Fish and Wildlife. Nature Conservancy of Canada, Victoria, British Columbia.

Jesinghaus, J. 1999. A European System of Environmental Pressure Indices, First Volume of the Environmental Pressure Indices Handbook: The Indicators. European Commission, Joint Research Centre, Ispra, Italy. Accessed May 2012 <http://esl.jrc.it/envind/theory/handb_.htm>.

Johnson, R. E., and K. M. Cassidy. 1997. Mammals of Washington state: location data and modeled distributions. Volume 3 in Washington State Gap Analysis - Final Report. Washington Cooperative Fish and Wildlife Research Unit, Seattle, Washington.

Karr, J.R. 1990. Biological integrity and the goal of environmental legislation: lessons for conservation biology. Conservation Biology 4:244-250.

Karr, J.R. 1991. Biological integrity: a long neglected aspect of water resource management. Ecological Applications 1:66-84.

Kerr, J.T. 1997. Species richness, endemism, and the choice of areas for conservation. Conservation Biology 11:1094-1100.

King, R.S., M.E. Baker, D.F. Whigham, D.E. Weller, T.E. Jordan, P.F. Kazyak, and M.K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15:137-153

Kruckeberg, A.R. 1991. The Natural History of Puget Sound Country. University of Washington Press, Seattle, Washington.

Lee, Se-Yeun, and A.F. Hamlet. 2011. Skagit River Basin Climate Science Report. Department of Civil and Environmental Engineering and the Climate Impacts Group, University of Washington, Seattle.

Lestelle, L.C., L.E. Mobrand, W.E. McConnaha. 2004. Information structure of ecosystem diagnosis and treatment (EDT) and habitat rating rules for Chinook salmon, coho salmon, and steelhead trout. Mobrand Biometrics Inc., Vashon Island, Washington.

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.

Marzluff, J.M., R. Bowman, and R. Donnelly. 2001. A historical perspective on urban bird research: trends, terms, and approaches. pp. 1-17 in J. M. Marzluff, R. Bowman, and R. Donnelly (eds.), Avian ecology and conservation in an urbanizing world. Kluwer Academic Press, Norwell, Massachusetts.

Mattson, K.M., and P.L. Angermeier. 2007. Integrating human impacts and ecological integrity into a risk-based protocol for conservation planning. Environmental Management 39:125-138.

May, C.W., and G. Peterson. 2003. Landscape assessment and conservation prioritization of freshwater and nearshore salmonid habitat in Kitsap County. prepared for Kitsap County, Port Orchard, Washington.

McElhany, P., E.A. Steel, K. Avery, N. Yoder, C. Busack, and B. Thompson. 2010. Dealing with uncertainty in ecosystem models: lessons from a complex salmon model. Ecological Applications 20:465-482.

McGarigal, K., and B.J .Marks. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. General Technical Report PNW-351. Pacific Northwest Research Station, USDA Forest Service, Portland, Oregon.

McKinney, M.L. 2002. Urbanization, biodiversity, and conservation. Bioscience 52:883-890.

Mebane, C.A., T.R. Maret, and R.M. Hughes. 2003. An index of biological integrity (IBI) for Pacific Northwest rivers. Transactions of the American Fisheries Society 132:239-261.

Meyer, J.L., D.L. Strayer, J.B.Wallace, S.L. Eggert, G.S. Helfman, and N.E. Leonard. 2007. The contribution of headwater streams to biodiversity in river networks. Journal of the American Water Resources Association 43:86-103.

Miller, D.J. 2003. Programs for DEM analysis. In Landscape Dynamics and Forest Management. RMRS-GTR-101-CD. Rocky Mountain Research Station, USDA Forest Service, Fort Collins, Colorado.

Mills, L.S., M.E. Soule, and D.F. Doak. 1993. The keystone-species concept in ecology and conservation. Bioscience 43:219-224.

Minshall, G.W., K.W. Cummins, R.C. Petersen, C.E. Cushing, D.A. Bruns, J.R. Sedell, and R.L. Vannote. 1985. Developments in stream ecosystem theory. Canadian Journal of Fisheries and Aquatic Sciences 42:1045-1055.

Morrison, M.L., B.G. Marcot, and R.W. Mannan. 1992. Wildlife-Habitat Relationships: Concepts and Applications. University of Wisconsin Press, Madison, Wisconsin.

Niemeijer, D. 2002. Developing indicators for environmental policy: data-driven and theory-driven approaches examined by example. Environmental Science and Policy 5:91-103.

Noss R.F. 1987. From plant communities to landscapes in conservation inventories: a look at the Nature Conservancy (USA). Biological Conservation 41:11-37.

NOAA. 2009. DRAFT Lower Duwamish River NRDA Programmatic Restoration Plan & Programmatic Environmental Impact Statement. National Oceanic and Atmospheric Administration and U.S. Fish and Wildlife Service, Seattle, Washington.

NWIFC. 2009. SSHIAP catchment data layer. Salmon and Steelhead Habitat Inventory and Assessment Program, Northwest Indian Fisheries Commission, Olympia, Washington.

Nysewander, D.R., J.R. Evenson. B.L. Murphie, and T.A. Cyra. 2005. Report Of Marine Bird and Marine Mammal Component, Puget Sound Ambient Monitoring Program for July 1992 to December 1999 Period. Wildlife Management Program, Washington State Department of Fish and Wildlife, Olympia, Washington.

Olden, J.D. 2003. A species-specific approach to modeling biological communities and its potential for conservation. Conservation Biology 17:854-863.

OFM. 2007. Washington state growth management population projections for counties: 2000 to 2030. Office of Financial Management, Olympia, Washington. http://www.ofm.wa.gov/pop/gma/projections07.asp

Orme, C.D.L., R.G. Davies, M. Burgess, et al. 2005. Global hotspots of species richness are not congruent with endemism or threat. Nature 436:1016–1019.

Palazzi, D., and P. Bloch. 2006. Priority Marine Sites for Conservation in the Puget Sound. Aquatic Resources Division, Washington Department of Natural Resources, Olympia, Washington.

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

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

Pearce, J., and S. Ferrier. 2001. The practical value of modelling relative abundance of species for regional conservation planning: a case study. Biological Conservation 98:33-43.

Pess, G.R., D.R. Montgomery, E.A. Steel, R.E. Bilby, B.E. Feist, and H.M. Greenberg. 2002. Landscape characteristics, land use, and coho salmon (*Oncorhynchus kisutch*) abundance, Snohomish River, Wash., U.S.A. Canadian Journal of Fisheries and Aquatic Sciences 59:613-623.

Pianka, E.R. 1988. Evolutionary Ecology (4th ed.). Harper & Row Publishers, New York, New York.

Pierce County. 2004. Salmon Habitat Protection and Restoration Strategy: Puyallup (WRIA 10) and Chambers/Clover Creek (WRIA 12) Watersheds. Pierce County, Tacoma, Washington.

Pierce, K. 2011. Final Report on High Resolution Change Detection Project. unpublished report. Habitat Program, Washington Dept. of Fish and Wildlife, Olympia, Washington.

Poiani, K.A., M.D. Merrill, and K.A. Chapman. 2001. Identifying conservation-priority areas in a fragmented Minnesota landscape based on the umbrella species concept and selection of large patches of natural vegetation. Conservation Biology 15:513-522.

Popper, K., G. Wilhere, and 10 other authors. 2007. The East Cascades – Modoc Plateau and West Cascades Ecoregional Assessments. Prepared by The Nature Conservancy and the Washington Department of Fish and Wildlife. The Nature Conservancy, Portland, Oregon.

Prendergast, J.R., R.M. Quinn, J.H. Lawton, B.C. Eversham, and D.W. Gibbons. 1993. Rare species, the coincidence of diversity hotspots and conservation strategies. Nature 365:335-337.

Quigley, T.M., R.W. Haynes, and W.J. Hann. 2001. Estimating ecological integrity in the interior Columbia River basin. Forest Ecology and Management 153:161-178.

R Development Core Team (RDCT). 2005. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0. URL http://www.R-project.org.

Ripley, T., G. Scrimgeour, and M.S Boyce. 2005. Bull trout (*Salvelinus confluentus*) occurrence and abundance influenced by cumulative industrial developments in a Canadian boreal forest watershed. Canadian Journal of Fisheries and Aquatic Sciences 62:2431-2442.

Roberge, J.M., and P. Angelstam. 2004. Usefulness of the umbrella species concept as a conservation tool. Conservation Biology 18:76-85.

Root, K.V., H.R. Akçakaya, and L. Ginzburg. 2003. A multispecies approach to ecological valuation and conservation. Conservation Biology 17:96-206

RTI. 2011. Washington State Parcel Database. Rural Technology Initiative, University of Washington, Seattle, Washington. http://depts.washington.edu/wagis/projects/parcels/

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.

Rykiel, E.J. 1996. Testing ecological models: the meaning of validation. Ecological Modelling 90:229-244.

Salmon Recovery Science Review Panel. 2000. Report from the SRSRP Meeting held Dec. 4–6. www.nwfsc.noaa.gov/trt/rsrp_docs/rsrpdoc2.pdf.

Sanborn. 2007. Gap Zone 1 Vegetation mapping final report. Prepared for the US Geological Survey GAP Program. Sanborn Inc., Portland, Oregon.

Saunders, D.A., R.J. Hobbs, and C.R. Margules. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5:18-32.

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.

Schlenger, P., A. MacLennan, E. Iverson, K. Fresh, C. Tanner, B. Lyons, S. Todd, R. Carman, D. Myers, S. Campbell, and A. Wick. 2011. Strategic Needs Assessment Report. Technical Report 2011-02. Published by Washington Department of Fish and Wildlife, Olympia, Washington, and U.S. Army Corps of Engineers, Seattle, Washington.

Schmolke, A., P. Thorbek, D.L. DeAngelis, and V. Grimm. 2010. Ecological models supporting environmental decision making: a strategy for the future. Trends in Ecology and Evolution 25:479-486.

Sheer, M.B., D.S. Busch, E. Gilbert, J.M. Bayer, S. Lanigan, J.L. Schei, K. Burnett, and D. Miller. 2009. Development and management of fish intrinsic potential data and methodologies: state of the IP 2008 summary report. Pacific Northwest Aquatic Monitoring Partnership Series 2009-004. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Seattle, Washington.

Shipman, H. 2008. A Geomorphic Classification of Puget Sound Nearshore Landforms. Puget Sound Nearshore Partnership Technical Report No. 2008-01. Seattle District, U.S. Army Corps of Engineers, Seattle, Washington.

Simenstad, C., M. Logsdon, K. Fresh, H. Shipman, M. Dethier, and J. Newton. 2006a. Conceptual Model for Assessing Restoration of Puget Sound Nearshore Ecosystems. Puget Sound Nearshore Partnership Technical Report No. 2006-03. Published by Washington Sea Grant Program, University of Washington, Seattle, Washington.

Simenstad, C.A., M. Ramirez, J. Burke, M. Logsdon, H. Shipman, C. Tanner, J. Toft, B. Craig, 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. 2011. Historical Change of Puget Sound Shorelines: Puget Sound Nearshore Ecosystem Project Change Analysis. Technical Report No. 2011-01. Washington Department of Fish and Wildlife, Olympia, Washington, and U.S. Army Corps of Engineers, Seattle, Washington.

Simenstad, C., D. Reed, and M. Ford. 2006b. When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. Ecological Engineering 26:27-39.

Simenstad, C., C.D. Tanner, R.M. Thom, and L.L. Conquest. 1991. Estuarine Habitat Assessment Protocol. prepared for U.S. Environmental Protection Agency, Region 10, Seattle, Washington.

Skidmore, P.B. 2006. An Assessment of Freshwater Systems in Washington State. The Nature Conservancy, Seattle, Washington.

Snohomish County. 2005. Snohomish River Basin Salmon Conservation Plan. Snohomish Basin Salmon Recovery Forum and Snohomish County Department of Public Works, Everett, Washington.

Stanley, S., S. Grigsby, T. Hruby, and P. Olson. 2010. Puget Sound Watershed Characterization Project: Description of Methods, Models and Analysis. Ecology Publication #10-06-05. Washington State Department of Ecology, Olympia, Washington.

Stanley, S., S. Grigsby, D. Booth, D. Hartley, R. Horner, T. Hruby, J. Thomas, P. Bissonnette, J. Lee, R. Fuerstenberg, P. Olson, and G. Wilhere. 2011. Puget Sound Characterization, Volume 1: The Water Resources Assessments. Ecology Publication #11-06-016. Washington Department of Ecology, Olympia, Washington.

Steel, E.A., B.E. Feist, D.W. Jensen, G.R. Pess, M.B. Sheer, J.B. Brauner, and R.E. Bilby. 2004. Landscape models to understand steelhead (*Oncorhynchus mykiss*) distribution and help prioritize barrier removals in the Willamette basin, Oregon, USA. Canadian Journal Fisheries and Aquatic Sciences 61: 999-101.

Theobald, D.M. 2010. Estimating natural landscape changes from 1992 to 2030 in the conterminous US. Landscape Ecology 25:999-1011.

Theobold, D.M., D.M. Merritt, and J.B. Norman. 2010. Assessment of threats to riparian ecosystems in the western U.S. U.S.D.A. Stream Systems Technology Center and Colorado State University, Fort Collins, Colorado.

Turnhout, E., M. Hisschemoller, and H. Eijsackers. 2007. Ecological indicators: between the two fires of science and policy. Ecological Indicators 7:215-228.

USDA. 2008. National Forest Inventoried Roadless Areas, final edition. Geospatial Service and Technology Center, U.S. Forest Service, Salt Lake City, Utah.

USFWS. 1980. Habitat Evaluation Procedures (HEP). ESM 102. U.S. Fish and Wildlife Service, Department of Interior, Washington, D.C.

USFWS. 1981. Standards for the Development of Habitat Suitability Index Models. ESM 103. U.S. Fish and Wildlife Service, Department of Interior, Washington, D.C.

USFWS. 1989. National Wetlands Inventory. U.S. Department of the Interior, Fish and Wildlife Service, Washington, D.C. http://www.fws.gov/wetlands/.

USFWS. 2005. Nisqually National Wildlife Refuge Comprehensive Conservation Plan. U.S. Fish and Wildlife Service, Nisqually National Wildlife Refuge Complex, Olympia, Washington.

Vander Schaaf, D., G. Wilhere, and 10 other authors. 2006. A Conservation Assessment of the Pacific Northwest Coast Ecoregion. Prepared by the Nature Conservancy, the Nature Conservancy of Canada, and Washington Department of Fish and Wildlife. The Nature Conservancy, Portland, Oregon.

Vannote, R. L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Wahl, T.R., B. Tweit, and S.G. Mlodinow. 2005. Birds of Washington: status and distribution. Oregon State University press, Corvallis, Oregon.

Wang L., P.W. Seelbach, and R.M. Hughes. 2006. Introduction to landscape influences on stream habitats and biological assemblages. American Fisheries Society Symposium 48:1-23.

Washington Wildlife Habitat Connectivity Working Group (WHCWG). 2010. Washington Connected Landscapes Project: Statewide Analysis. Washington Department of Fish and Wildlife and Department of Transportation, Olympia, Washington.

Watson, J.W., and E.A. Rodrick. 2000. Bald Eagle. pp. 9-1 – 9-15 in E.M. Larsen, J.M. Azerrad, and N. Nordstrom (tech. eds.). Management Recommendations for Washington's Priority Species − Volume IV: Birds. Washington Dept. of Fish and Wildlife, Olympia, Washington.

WDF. 1992. Salmon, Marine Fish, and Shellfish Resources and Associated Fisheries in Washington's Coastal and Inland Marine Waters. Technical Report 79 revised. State of Washington Department of Fisheries, Olympia, Washington.

WDFW. 2008a. Priority Habitat and Species List. Washington Department of Fish and Wildlife, Olympia, Washington.

WDFW. 2008b. Deschutes Estuary Feasibility Study Final Report. Prepared by Philip Williams & Associates, Ltd. with ECONorthwest and AMEC Earth & Environmental, Inc. for the Washington Dept. of Fish and Wildlife, Olympia, Washington.

WDFW and WWTIT. 2002. Salmonid Stock Inventory (SaSI). Washington Department of Fish & Wildlife and the Western Washington Treaty Indian Tribes, Olympia, Washington.

Western Governors' Wildlife Council (WGWC). 2011. Western Wildlife Crucial Habitat Assessment Tool (CHAT):Vision, Definitions and Guidance for State Systems and Regional Viewer. Western Governors' Association, Denver, Colorado.

Wilhere, G.F. 2012. Using Bayesian networks to incorporate uncertainty in habitat suitability index models. Journal of Wildlife Management 76:1298-1309.

Wilhere, G.F. 2008. The how-much-is-enough myth. Conservation Biology 22:514-517.

Wilhere, G.F., M. Goering, and H. Wang. 2008. Average optimacity: an index to guide site prioritization for biodiversity conservation. Biological Conservation 141:770-781.

Wilhere, G.F., M.J. Linders, and B.L. Cosentino. 2007. Defining alternative futures and projecting their effects on the spatial distribution of wildlife habitats. Landscape and Urban Planning 79:385-400.

Williams, P., D. Gibbons, C. Margules, A. Rebelo, C. Humphries, and R. Pressey. 1996. A comparison of richness hotspots, rarity hotspots, and complementary areas for conserving diversity of British birds. Conservation Biology 10:155-174.

Wilson, D.S. 1997. Biological communities as functionally organized units. Ecology 78:2018-2024.

Winston, M.R., and P.L. Angermeier. 1995. Assessing conservation values using centers of population density. Conservation Biology 9:1518-1527.

Wipfli, M.S., J.S. Richardson, and R.J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. Journal of the American Water Resources Association 43:72-85.

Worm, B. and J.E. Duffy. 2003. Biodiversity, productivity and stability in real food webs. Trends in Ecology and Evolution 18:628-632.

Wydoski, R.S., and R.R. Whitney. 2003. Inland Fishes of Washington. University of Washington Press, Seattle, Washington.

Appendices

Appendix A:
Methods for Terrestrial Habitats Assessment

The conservation value of AUs was calculated in three stages. In the first stage open-space blocks were identified. An open-space block is a contiguous area containing land uses – such as commercial forest, agriculture, parks, and designated open-space – that maintain natural or semi-natural habitats or serve as habitats for native wildlife. Second, the landscape integrity of open-space blocks was assessed. And third, the landscape integrity of open-space blocks was combined with PHS habitats found in AUs, including oak-grassland habitats, to yield each AU's relative value for conservation.

Open-Space Blocks

Open-space blocks are comprised of open-space parcels. Three spatial data layers were used to identify open-space parcels: the Washington State Parcel Database developed by the Rural Technology Initiative (RTI 2011), land cover data developed by WDFW (Pierce 2011) using aerial photography from the National Agriculture Imagery Program, and the Major Public Lands spatial data layer created by the Washington Department of Natural Resources (Figure A1).

WAC 458-53-030 lists 83 different land uses recognized under the Washington State tax code (Table A1). The Washington State Parcel Database contains the land use for all private land parcels in Puget Sound Basin. We grouped the 83 land uses into seven general categories: commercial-industrial, residential, agriculture, forestry, mining, mixed-use open space, designated open space. All categories except commercial-industrial can contribute to open space. For each general category we constructed rules that classified parcels as open space or not open space (Table A2). The rules where developed through an iterative process of constructing an initial rule, applying it to the parcel database, comparing the result against aerial photography to determine how accurately the rule identified open-space parcels, adjusting the rule to increase its accuracy, and repeating this process until a satisfactory level of accuracy was achieved. Rules consisted of three variables − parcel size, land cover, and vegetation zone – and adjustments to the rules were based on expert judgments. In the Washington State Parcel Database, data for state and federally managed public lands are missing or inconsistent. Consequently, for state and federally managed public lands we used the Major Public Lands spatial data layer. Department of Defense lands are often a mix of intensively developed areas and semi-natural open space. For large military installations (>1000 acres), we used aerial photography to identify and delineate developed areas and semi-natural open space.

The non-open space parcels were removed from the parcel database, and major highways (state routes, federal and interstate highways) were intersected with the remaining parcels which caused some parcels to be split into smaller polygons. Boundaries between adjacent parcels were dissolved to form larger polygons and only polygons greater than 10 acres were retained as the final set of open-space blocks.

Table A1. Relative impact values assigned to each land use. Land uses listed in WAC 458-53-030, “Stratification of assessment rolls - Real property.” Codes are found in the StateLandUseCode attribute of the Washington State Parcel Database (RTI 2011). Impact Values based on subjective professional judgment.

Code	Land Use Category	Min Impact Value	Max Impact Value
11	Household, single family units	100	1000
12	Household, 2-4 units	100	1000
13	Household, multiunits (5 or more)	100	1000
14	Residential condominiums	100	1000
15	Mobile home parks or courts	100	1000
16	Hotels/motels	1000	1000
17	Institutional lodging	1000	1000
18	All other residential not elsewhere coded	1000	1000
19	Vacation and cabin	1000	1000
21	Food and kindred products	1000	1000
22	Textile mill products	1000	1000
23	Apparel and other finished products made from fabrics, leather, and similar	1000	1000
24	Lumber and wood products (except furniture)	1000	1000
25	Furniture and fixtures	1000	1000
26	Paper and allied products	1000	1000
27	Printing and publishing	1000	1000
28	Chemicals	1000	1000
29	Petroleum refining and related industries	1000	1000
30	Rubber and miscellaneous plastic products	1000	1000
31	Leather and leather products	1000	1000
32	Stone, clay and glass products	1000	1000
33	Primary metal industries	1000	1000
34	Fabricated metal products	1000	1000
35	Professional scientific, and controlling instruments; photographic and optical	1000	1000
39	Miscellaneous manufacturing	1000	1000
41	Railroad/transit transportation	1000	1000
42	Motor vehicle transportation	1000	1000
43	Aircraft transportation	100	1000
44	Marine craft transportation	1000	1000
45	Highway and street right of way	1000	1000
46	Automobile parking	1000	1000
47	Communication	1000	1000
48	Utilities	100	1000
49	Other transportation, communication, and utilities not classified elsewhere	1000	1000
50	Condominiums - other than residential condominiums	1000	1000
51	Wholesale trade	1000	1000
52	Retail trade - building materials, hardware, and farm equipment	1000	1000
53	Retail trade - general merchandise	1000	1000
54	Retail trade - food	1000	1000
55	Retail trade - automotive, marine craft, aircraft, and accessories	1000	1000
56	Retail trade - apparel and accessories	1000	1000
57	Retail trade - furniture, home furnishings and equipment	1000	1000
58	Retail trade - eating and drinking	1000	1000
59	Other retail trade	1000	1000
61	Finance, insurance, and real estate services	1000	1000
62	Personal services	1000	1000
63	Business services	1000	1000
64	Repair services	1000	1000
65	Professional services	1000	1000
66	Contract construction services	1000	1000

Code	Land Use Category	Min Impact Value	Max Impact Value
67	Governmental services	100	1000
68	Educational services	100	1000
69	Miscellaneous services	100	1000
71	Cultural activities and nature exhibitions	100	1000
72	Public assembly	100	1000
73	Amusements	100	1000
74	Recreational activities	100	1000
75	Resorts and group camps	100	1000
76	Parks	100	1000
79	Other cultural, entertainment and recreational	100	100
81	Agriculture (not classified under current use law)	100	100
82	Agriculture related activities	100	100
83	Agriculture classified under current use chapter 84.34 RCW	100	100
84	Fishing activities and related services	100	100
85	Mining activities and related services	100	1000
88	Designated forest land under chapter 84.33 RCW	10	10
89	Other resource production	10	1000
91	Undeveloped land	10	1000
92	Noncommercial forest	10	10
93	Water areas	0	0
94	Open space land classified under chapter 84.34 RCW	10	10
95	Timberland classified under chapter 84.34 RCW	10	10
99	Other undeveloped land	10	1000

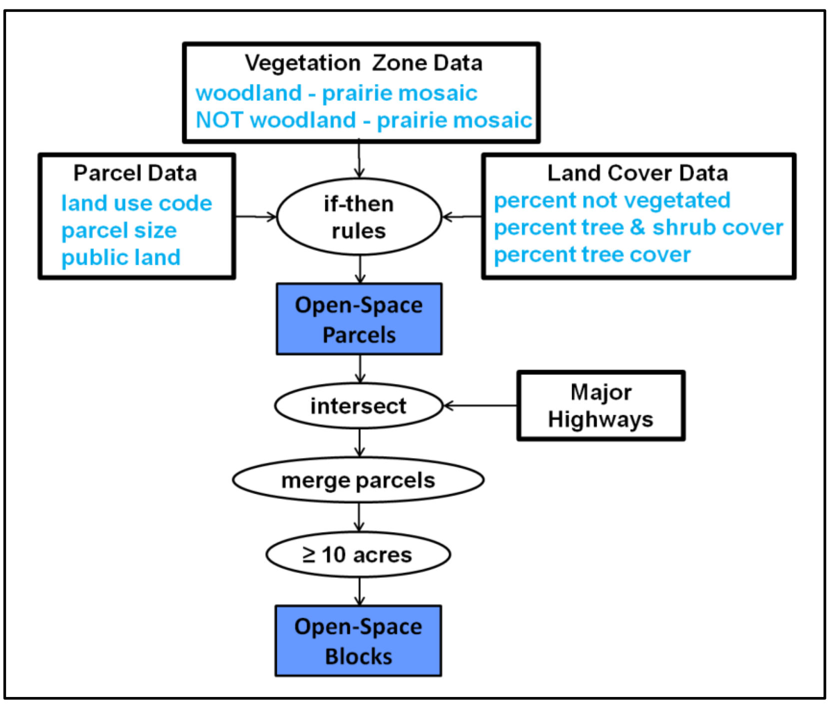


Figure A1. Process and spatial data used to construct open-space blocks. Land cover data were from Pierce (2011), parcel data were from the Rural Technology Initiative (RTI 2011), and vegetation zone data were modified from Cassidy (1997).

Table A2. Rules for identifying open space blocks using parcel layer, vegetation zones, and land cover.

Land Use Code[1]	Parcel Area (acres)	Vegetation Zone[2]	Land Cover[3]
land use ≤ 79 and not in [45, 19, 45, 48, 67, 68, 71, 72, 74, 75,76]	4.5 < Area ≤ 5	1	PR_NoVeg < 0.05
		≠ 1	(PR_NoVeg < 0.2) AND ((PR_High_Tr > 0.65) OR (PR_TreShrb > 0.75))
	5 < Area ≤ 10	1	PR_NoVeg < 0.05
		≠ 1	PR_NoVeg < 0.2) AND ((PR_High_Tr > 0.6) OR (PR_TreShrb > 0.7))
	10 < Area ≤ 40	1	PR_NoVeg < 0.05
		≠ 1	PR_NoVeg < 0.2) AND ((PR_High_Tr > 0.55) OR (PR_TreShrb > 0.6))
	40 < Area	1	PR_NoVeg < 0.05
	40 < Area ≤ 160	≠ 1	(PR_NoVeg < 0.2) AND ((PR_High_Tr > 0.5) OR ("PR_TreShrb > 0.55))
	Area < 160	--	PR_NoVeg < 0.3
land use in [19, 48, 67, 68, 71, 72, 74, 75, 76]	--	--	(PR_NoVeg < 0.1) AND ((PR_High_Tr > 0.25) OR (PR_TreShrb > 0.50))
81 ≤ land use ≤ 85	4.5 < Area ≤ 5	1	PR_NoVeg < 0.1
	5 < Area ≤ 10	--	PR_NoVeg < 0.15
	10 < Area ≤ 40		PR_NoVeg < 0.1
	40 < Area		PR_NoVeg < 0.05
(87 ≤ land use ≤ 99) and not 93	Area ≤ 5	--	(PR_NoVeg < 0.1) AND (PR_TreShrb > 0.2)
	5 < Area ≤ 10		(PR_NoVeg < 0.1) OR (PR_TreShrb > 0.4)
	10 < Area ≤ 40		(PR_NoVeg < 0.05) OR (PR_TreShrb > 0.3)
	40 < Area		--

[1] See Table A1 for meaning of land use codes

[2] vegetation zones: 1 equals woodland-prairie mosaic. Other major zones are all coniferous forest types.

[3] PR_NoVeg = proportion of parcel with no vegetation; PR_TreShub = proportion of parcel covered by trees and shrubs; PR_High_Tr = proportion or parcel covered by trees.

Landscape Integrity of Open Space Blocks

Ecological integrity is the ability of an ecological system to support and maintain a community of organisms that has species composition, diversity, and functional organization comparable to those of natural habitats within a region (Parrish et al. 2003). Ecological integrity is a multi-scale concept, and can be assessed at a stand (or site) scale, landscape scale, and scales in between. Because this is a regional assessment covering a huge spatial extent, we assessed only landscape-scale ecological integrity, which we henceforth refer to as *landscape integrity*. Our index of relative landscape integrity was based on expert judgment.

Land use and fragmentation have been demonstrated to affect ecological integrity (e.g., Glennan and Porter 2005). Hence, the *relative* landscape integrity of each open-space block was a function of land use impacts and open-space fragmentation (Figure A2). Three spatial data layers were used to assess landscape integrity: the Washington State Parcel Database (RTI 2011), land cover data developed by WDFW (Pierce 2011), and the open-space blocks described in the preceding section.

Land use impacts were a function of land use and land cover. We grouped the 83 private land uses into eight general categories: commercial-industrial, residential, agriculture, forestry, mining, mixed-use open space, designated open space, and public lands. All lands not in an open space block were assigned the highest relative impact regardless of land use or land cover (1000; Table A1). Relative impact of residential parcels and mixed-use open space parcels ranged from 100 to 1000 depending on the proportion of the parcel covered by grass, shrubs, and trees for parcels in the oak-prairie mosaic vegetation zone and on the proportion of the parcel covered by trees for parcels in all other vegetation zones. All other zones are coniferous forest zones. Because some parcels classified as forestry or open space can contain residences, the relative impacts of forestry parcels and designated open space parcels were also functions of land cover that ranged from 10 to 1000. State and federal public lands were assigned relative impact values (Table A2) based on expert judgment. We supplemented the Major Public Lands data with the National Forest Inventoried Roadless Areas spatial data layer (USDA 2008). Wilderness and roadless areas were assigned a relative impact value of 0.

The relative landscape integrity of an open-space block was in part a function of land use impacts inside the block and land use impacts surrounding the block. Land use impact inside each block was calculated as a weighted arithmetic mean of parcels' land use impacts within the block, where the weight was the area of each parcel. Land use impact outside each block was based on land use impacts at four distances from the block: 0 to 0.5, 0.5 to 1, 1 to 2, and 2 to 4 miles. An area weighted mean impact was calculated for each ring and the overall impact outside a block was a weighted mean of these four rings with the weights set to 12, 4, 2, and 1 (nearest to farthest ring).

Metrics of open-space fragmentation were calculated for each open-space block using the program FRAGSTATS (McGarigal and Marks 1995). The indices used in the landscape integrity index were the shape index called "Circle" and the proximity index (Gustafson and Parker 1994). The proximity index incorporates both patch isolation and patch density, which are factors influencing habitat connectivity. The search radius for the proximity index was 4 miles. All indices were normalized so that the maximum value equaled 1. Block size was normalized by dividing size by 50,000 acres (20,235 ha), the minimum size assumed to be necessary for fully intact landscape integrity. Proximity values were highly skewed to the right, hence, proximity was normalized by dividing all values by the 90^{th} percentile of proximity. The shape of a block and the condition of the surrounding landscape become less important as block size increases. Hence, the weights for block shape and external impacts were linear functions of block size. The equation for landscape integrity is shown schematically in Figure A2 and weights are summarized in Table A4.

See Box A1 for detailed description of operations used to calculate the landscape integrity index.

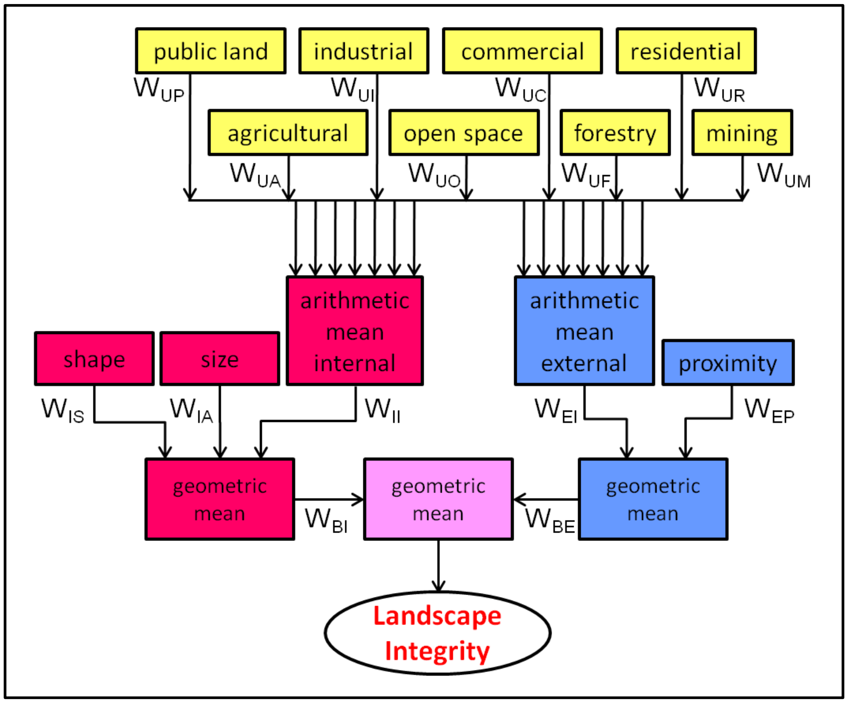


Figure A2. Structure of model used to create landscape integrity index. W_{xx} are weights, i.e. parameters, used in weighted geometric or arithmetic means. Proximity and shape indices were calculated with FragStats (McGarigal and Marks 1995) for each open space block.

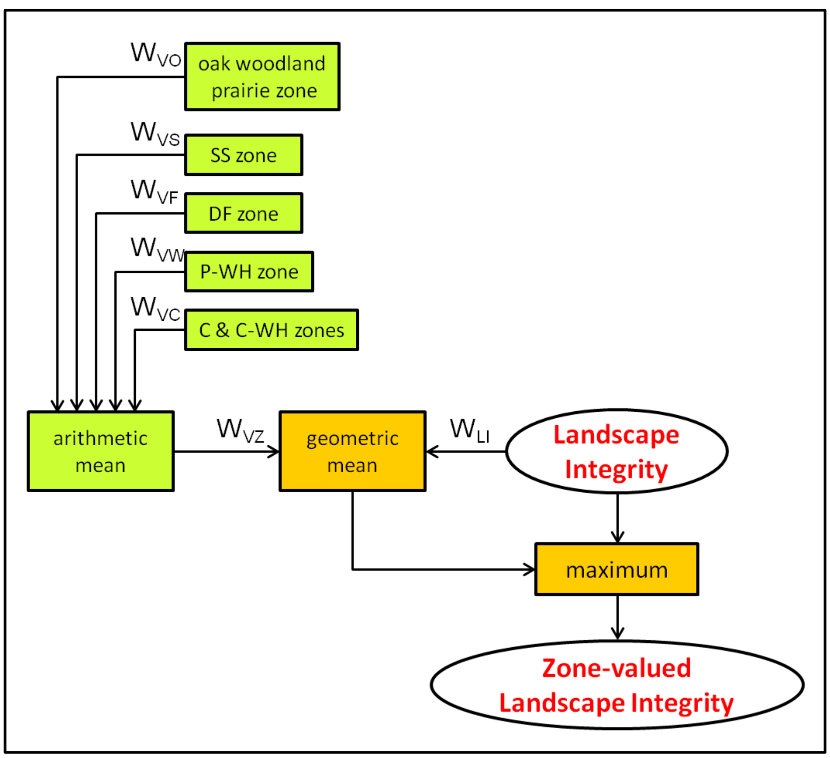


Figure A3. Combining vegetation zone index and landscape integrity index to yield overall zone-values integrity index. W_{xx} are weights, i.e., parameters, used in weighted geometric or arithmetic means. The maximum function does the following: if the landscape integrity index is high, then vegetation zone will not reduce it, but if integrity index is low, then vegetation zones can enhance it.

Table A3. Relative impact values assigned to public lands. Reference point is relative impact value assigned to timberland and designated forest land (Table A1). Impact Values based on subjective professional judgment.

Ownership / Management	Relative Impact Value
federal wilderness area	0
federal roadless area	0
national park	1
national wildlife refuge	4
national forest, non-wilderness & not roadless	4
municipal watersheds	8
Washington Dept of Natural Resources	8
Washington Dept. of Fish & Wildlife	10
Dept of Defense, open space	10
Washington State Parks	15

Table A4. Subjective weights used in calculation of landscape integrity index. Weights for open-space block shape and external characteristics are functions of open-space block size. Functions of block size are indicated by f(size) and g(size).

Symbol	Value	Name	Equation Schematic
W_{IA}	4	open-space block size	Figure A2
W_{IS}	f(size)	open-space block shape (range 0 to 1)	
W_{II}	2	open-space block internal impacts	
W_{EP}	1	open-space block proximity	Figure A2
W_{EI}	1	open-space block external impacts	
W_{VO}	20	oak woodland-prairie mosaic zone	Figure A3
W_{VF}	4	Puget Sound Douglas-fir zone	
W_{VW}	2	Puget Trough western hemlock	
W_{VC}	1	Cascades western hemlock zone	
W_{VS}	3	Sitka spruce zone	
W_{BI}	2	internal characteristics	Figure A2
W_{BE}	g(size)	external characteristics (range 0 to 1)	
W_{VZ}	10	vegetation zones	Figure A3
W_{LI}	1	landscape integrity	

Vegetation zone was also a factor used to influence the value of open-space blocks. Our vegetation zones were based on the GAP vegetation zones (Cassidy 1997) which we modified. We revised the Oak Woodland-Prairie Mosaic zone by using the 2005 oak woodland and grassland prairie data of the Washington Natural Heritage Program, historical prairie data from Robert Van Pelt at the University of Washington, and a mapping of dry prairie soils from the Natural Resources Conservation Service. These data were combined and edited to map a reasonable potential distribution of oak woodlands and prairies throughout the Puget Sound Basin. The Western Hemlock Zone was split using ecoregion boundaries to make a distinction between lower and higher elevation forests. Portions of the Western

Hemlock Zone in the Puget Trough Ecoregion were renamed Puget Trough Western Hemlock. In the North Cascades and West Cascades Ecoregions, the Western Hemlock Zone were renamed Cascades Western Hemlock, and in the Northwest Coast Ecoregion the higher elevation Western Hemlock zone was renamed Coastal Western Hemlock. For each open-space block, an average vegetation zone value was calculated based on the area of each vegetation zone intersecting the open-space block and the relative value assigned to each vegetation zone. The relative value of each vegetation zone was a subjective judgment based on the percent of historical area lost and rarity of the zone. The equation for combining landscape integrity and vegetation zones to yield a *zone-valued landscape integrity* is shown schematically in Figure A3 and weights are summarized in Table A4.

Box A1. Pseudo-code describing the algorithm used to calculate terrestrial integrity index. Actual code was implemented in ArcGIS with data table selection queries and field calculator.

```
Set REL_Impact = 1000 for all records, then recalculate REL_Impact for following LandUse codes as follows:

# Commercial and industrial land uses; 249=Dept of Corrections, 45=Highways
If LandUse in (16, 17, 18, 19, . . . , 41, 42, 44, 45, 46, 47, 49, 50, 51, . . ., 66, 249) Then Rel_Impact = 1000

#----------------------------------------------------------------------------------------------------------------------------------
# residential and recreational land uses
# not in Woodland/Prairie Mosaic Zone
If ((LandUse <=15) OR (LandUse in (43, 48, 67, 68, 69, 70, 71, 72, 73, 74, 75, 76, 77, 78, 79, 85)) )
AND (veg_zone>1) Then
    Rel_Impact = -900*[PR_High_Tr] + 1000              # relative impact score between 100 and 1000

# in Woodland/Prairie Mosaic Zone
if ((LandUse<=15) OR (LandUse in (43, 48, 67, 68, 69, 70, 71, 72, 73, 74, 75, 76, 77, 78, 79, 85)) )
AND (veg_zone=1) Then
    Rel_Impact = -900*(0.33*[PR_Low_Gra] + [PR_TreShrb]) + 1000        # [PR_Low_Gra] + [PR_TreShrb] < 1

#----------------------------------------------------------------------------------------------------------------------------------
# undeveloped land use and not presently assigned codes (just to be safe)
# not in Woodland/Prairie Mosaic Zone
If (LandUse in (86, 89, 91, 96, 97, 98, 99)) AND (veg_zone>1) Then
Rel_Impact = -990*[PR_TreShrb] + 1000                  # relative impact score between 10 and 1000

# in Woodland/Prairie Mosaic Zone
If (LandUse in (86, 89, 91, 96, 97, 98, 99)) AND (veg_zone=1) Then
REl_Impact = -990*(0.67*[PR_Low_Gra] + [PR_TreShrb]) + 1000            # score between 10 and 1000
#----------------------------------------------------------------------------------------------------------------------------------

If LandUse in (81, 82, 83, 84)  Then REL_Impact = 100         # agriculture

# private timberland, commercial forest, WDFW (221, 222), and open space on Dept of Defense (341)
If LandUse in (87, 88, 92, 94, 95, 221, 222, 341) Then REL_Impact = 10
If LandUse in (101, 211, 351) Then REL_Impact = 8         #municipal watersheds, DNR, BLM
If LandUse in (311, 335) Then REL_Impact = 4              #USFWS, National Forest- recreation or undesignated
If LandUse in (231, 322) Then REL_Impact = 15             #WA State Parks, National Park- historic park
If LandUse = 321 Then REL_Impact = 1                      #National Parks
If LandUse in (93, 331, 332) Then REL_Impact = 0          #water, federal wilderness, federal roadless areas
```

Species and Habitat Based Indices

Places with high zoned-valued landscape integrity do not necessarily coincide with the locations of priority habitat and species. Hence, priority habitat and species (PHS) data were incorporated into the index. However, much of the PHS data are site-scale (e.g., nest and den sites), which does not match the scale of the assessment. Site-scale occurrences are best addressed by site-level management. This assessment is intended for landscape-scale or regional land use planning. Hence, we used only PHS data that were landscape-scale occurrences, defined as occurrences greater than 10 to 100 acres in size, depending on the species (Table A5). We also wanted data that were accurate, i.e., data with a low rate of false negatives, and therefore, represented nearly all habitat (> 85%) for that species in the Puget Sound Basin. These two filters limited the PHS data to 12 species represented by 441 polygons ranging in size from 10 to 1 million acres. The smallest polygons were for Taylor's checkerspot butterfly and the largest polygons were for elk (Figure A5).

We also included two PHS habitat types, Oregon white oak woodlands and westside prairie, which were lumped into one habitat type − an oak-grassland type. The oak-grassland habitat type is perhaps the most imperiled terrestrial habitat type in the Puget Sound Basin. The Washington Natural Heritage Program has mapped prairie and oak woodland types with about the same degree of accuracy and precision as the PHS data we included in our assessment. Hence, we included the Heritage Program's oak-grassland habitat data and treated it in the same way as PHS data.

We developed a simple index for PHS habitats. For each AU we calculated for all 12 species the percent of the AU covered by the PHS polygons of each species and the percent of each species' entire habitat in the Basin contained within the AU (Figure A6). These 24 numbers were then adjusted such that a percentage greater than a threshold, T, was set to 100 and percentages less than T were translated to a 0 to 100 scale. T was set to 25% for all PHS habitats except elk and waterfowl concentrations, for which T was set to 50%. The rationale for this threshold is that an AU that is greater than 25% PHS habitats, for instance, or contains more than 25% of a species' habitat in the Basin is invaluable. The index was the maximum of these 24 adjusted percentages. The same process was applied to The Audubon Society's Important Bird Areas and the oak-grassland habitat types.

Table A5. Summary of spatial data from WDFW's Priority Habitats and Species (PHS) database, the Washington Natural Heritage Program (WNHP), and the Audubon Society's Important Bird Areas (IBAs) that were used in the terrestrial habitats assessment. All areas in acres. See Figure A5 for map of PHS polygons and IBAs.

Feature	**number of polygons**	**smallest size**	**largest size**	**mean size**	**total area**
WDFW PHS					
Taylor's checkerspot butterfly	6	10	140	45	270
streaked horned lark	1	670	670	670	670
sandhill crane	1	1,040	1,040	1,040	1,040
swan overwintering	41	20	3,210	400	16,495
bald eagle communal roosts	96	10	12,210	330	31,740
peregrine falcon overwintering	4	680	11,990	6,100	24,390
harlequin duck	99	10	4,900	290	29,030
waterfowl concentrations	120	100	28,410	1025	125,090
shorebird concentrations[1]	28	10	600	150	4,090
seabird concentrations[1]	4	30	680	310	1,245
Mazama pocket gopher	17	10	620	175	2,980
elk	15	12	1,058,170	145,880	2,188,260
WNHP					
oak woodland and grasslands	1,238	0.1	3,780	19	24,070
Audubon					
Important Bird Areas	9	290	6,780	1,810	16,285

[1] Shorebird concentrations, seabird concentrations, and Important Bird Areas only included where they overlap the terrestrial assessment units.

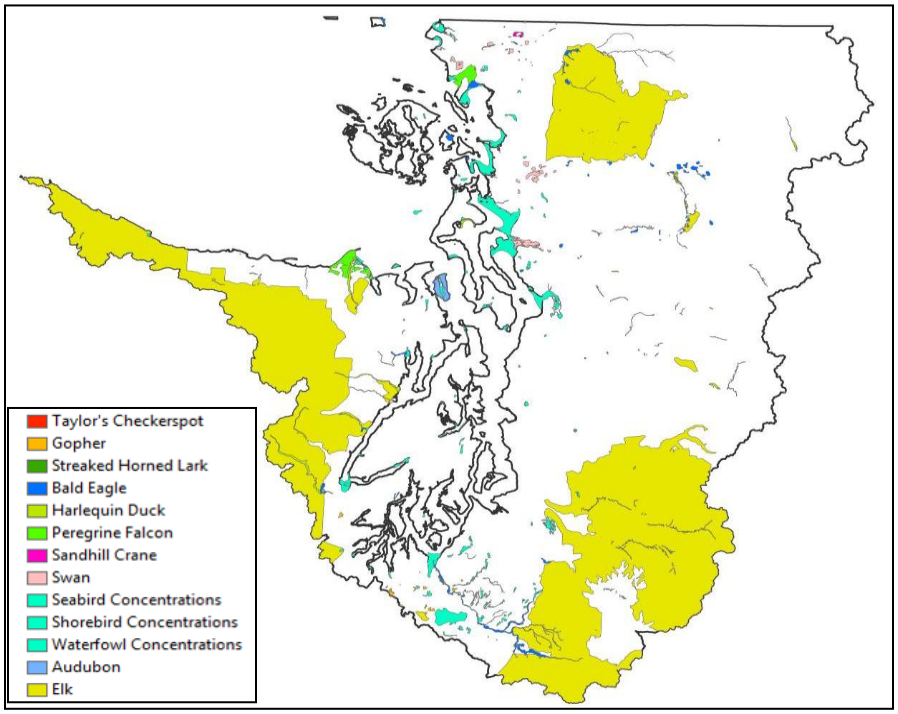


Figure A5. Polygons from Priority Species and Habitats (PHS) database and Audubon's Important Bird Areas used in the assessment. There are 441 polygons, not including the Important Bird Areas. See Table A3 for more information of PHS data.

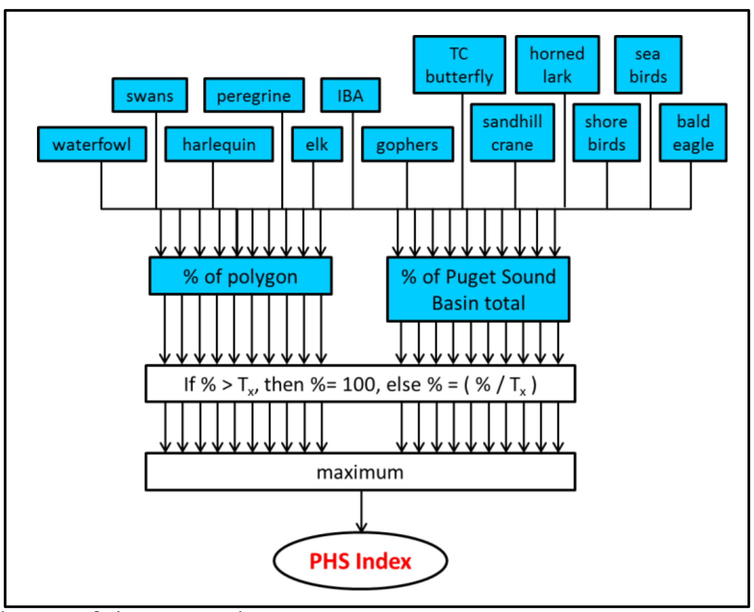


Figure A6. Calculation of the PHS Index. T_X represents 14 parameters – separate thresholds for each species or species group. T_X was set to 25% for all species except elk and waterfowl, for which T_X was set to 50%. Those values were based on professional judgment. The same process was applied to the oak-prairie habitats layer. IBA refers to Audubon's Important Bird Areas, TC butterfly means Taylor's checkerspot butterfly, and horned lark refers to the streaked horned lark. "Gophers" refers to the various subspecies of Mazama pocket gophers. Waterfowl, sea bird, and shorebird refer to concentration areas for a variety of bird species.

Relative Conservation Value of Assessment Units

We created one index of relative conservation value that was a function of three components: mean zone-valued landscape integrity index, the PHS index, and oak-grassland habitats index. The function was simply the maximum of the three components.

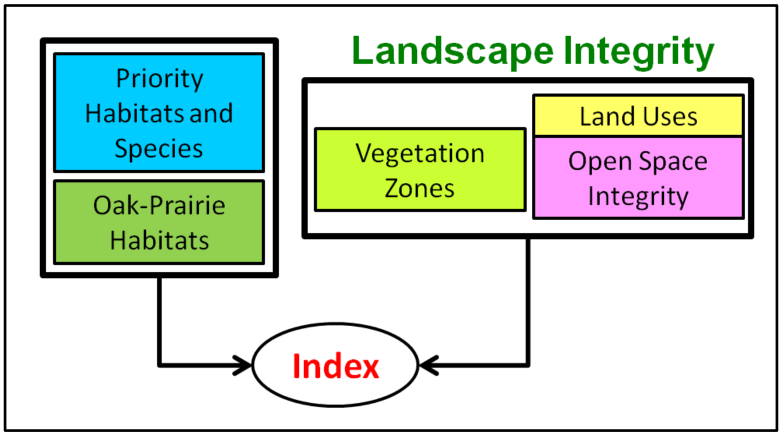


Figure A4. Major components of the terrestrial habitats assessment for the Puget Sound Basin. The relative conservation value index is calculated for each AU. Zone-valued landscape integrity is calculated for open-space blocks and then averaged within AUs.

Sensitivity and Uncertainty Analysis

We conducted both sensitivity and uncertainty analyses on the model (see Part 1). Sensitivity analysis was done by calculating the index for all AUs with the parameter values shown in Figures A2, A3, and A6, recalculating the index after altering a single parameter by a small amount (e.g., 5%), and applying equation 1.2 to each AU. The process was repeated for each parameter. Sensitivity analysis was done for 28 parameters, which does not include the land use impact values in Table A1.

Uncertainty analysis indicates the degree of confidence we should have in the model (see Part 1). Uncertainty analysis was done by assigning a uniform probability distribution to each of 28 parameters in Figure A2, A3, and A6. The distributions spanned the range of reasonable values for each parameter. Over 100,000 iterations, a parameter value was randomly selected from each distribution and the index was calculated for each AU. The index value of each AU is effectively a separate model output, and hence each AU has its own distribution of index values. The uncertainty associated with each AUs index value was calculated and depicted as a 90% confidence interval.

Appendix B: Methods for Lotic Freshwater Habitats Assessment

Analysis Units

Our assessment calculates a spatially-explicit index that indicates the relative conservation value of places throughout the Puget Sound Basin. The "places" are small watersheds (Table B1) which we call analysis units (AUs). The AUs are the same AUs used by the Department of Ecology for their assessments of water resources, i.e., water flow and water quality (Stanley et al. 2010).

AUs were derived from reach-scale catchments delineated by the Salmon and Steelhead Habitat Inventory and Assessment Program (SSHIAP; NWIFC 2009). The SSHIAP stream reach segmentation is based on channel gradient and channel confinement. The corresponding catchments are a very small size, with some encompassing only 0.01 square miles (64 acres). Ecology and WDFW believed that the water resources and habitats assessments could not be accurate at this scale due to the resolution of some spatial data layers used for the assessments (i.e. 1:24,000 and smaller). We thought AUs on the order of 1 to 10 square miles were more reasonable. Consequently, the SSHIAP catchments were aggregated into larger analysis units. These aggregations were assembled based on similar landform, geologic, and water flow characteristics. Two-thousand nine-hundred forty AUs were created. Sizes ranged from 0.004 to 21.8 square miles, with median and mean sizes of 3.5 and 4.7 square miles, respectively.

Table B1. Summary of analysis unit sizes by WRIA. Units in square miles. For purposes of this analysis the two Skagit River WRIAs (3 and 4) were lumped into a single WRIA.

WRIA	N	Mean	min	1st qtr	median	3rd qtr	max
1	247	5.0	0.06	1.3	4.4	8.0	16.6
2	154	1.1	0.004	0.6	1.0	1.4	4.8
3 & 4	421	7.2	0.29	4.1	7.5	10.1	18.4
5	107	6.6	0.32	3.7	6.0	9.5	16.7
6	124	1.7	0.02	0.8	1.2	2.0	7.9
7	268	7.0	0.16	3.9	6.4	9.9	17.7
8	139	4.5	0.01	2.5	4.1	5.7	16.7
9	95	5.7	0.63	2.3	4.5	9.1	16.1
10	151	6.9	0.63	3.9	6.7	9.6	15.1
11	109	6.9	0.65	3.8	6.0	9.6	16.8
12	60	2.9	0.66	1.8	2.6	3.8	6.6
13	83	3.2	0.48	1.1	2.3	4.2	15.7
14	167	2.0	0.15	0.8	1.1	3.2	8.4
15	341	1.9	0.29	1.0	1.5	2.2	10.0
16	106	5.7	0.12	1.0	4.8	10.4	21.8
17	169	2.4	0.33	0.8	1.0	2.8	14.5
18	103	6.8	0.27	2.3	5.9	10.5	20.6
19	96	4.0	0.38	1.1	3.0	4.9	19.8
Basin	2940	4.7	0.004	1.2	3.5	7.3	21.8

AUs were the spatial grain of our assessment. The spatial extent of our assessment was, in effect, each individual WRIA. That is, the freshwater habitats assessment was comprised of 18 separate assessments each corresponding to a WRIA[19]. Relative conservation value was relative within a WRIA, and hence, it cannot be used to compare the conservation value of AUs in different WRIAs. However, the spatial data and assessment methods were consistent across WRIAs, and therefore, the assessment can be used to discern patterns in relative conservation value among WRIAs.

Stream Reaches

Some components of our index needed specific information on individual river and stream reaches: length, channel gradient, active channel width, valley floor width, and mean annual flow. No currently available spatial data layers provided such information, so we contracted with M2 Environmental Services to create one. Their NetTrace software (Miller 2003, Clarke et al. 2008) uses digital elevation models (DEMs) to generate a routed channel network. The resulting channel network for the entire Puget Sound Basin had 706,744 stream reaches with an average length of 117 meters (Table B2). The mean, median, and range of reach lengths were remarkably consistent across all WRIAs. On average, there were 240 stream reaches per AU. The hydrography generated by NetTrace was compared to existing hydrography (i.e., DNR 2006) and was found to be highly congruent.

Table B2. Summary of stream reaches generated with NetTrace (Miller 2003). Units in meters. For purposes of this analysis the two Skagit River WRIAs (3 and 4) were lumped into a single WRIA.

WRIA	N	mean	min	1st qtr	median	3rd qtr	max
1	69,243	117	21	104	107	110	1014
2	2,529	115	86	105	108	114	437
3 & 4	193,727	115	14	104	107	110	1014
5	35,962	120	20	104	107	111	1015
6	2,100	117	69	105	108	115	331
7	114,452	117	10	104	107	110	1014
8	19,023	123	37	104	107	112	1014
9	24,234	121	23	104	107	111	1014
10	57,313	118	9	104	107	111	1014
11	40,334	120	23	104	107	111	1014
12	1,365	165	100	106	114	143	1009
13	10,749	120	56	104	107	111	1013
14	9,040	124	57	104	108	113	1011
15	16,487	118	31	105	108	113	1007
16	31,346	117	41	104	107	110	1014
17	15,939	116	15	104	107	111	1012
18	40,291	117	21	104	107	110	1013
19	22,610	116	42	104	107	110	1012
ALL	706,744	117	9	104	107	110	1015

[19] For the purposes of this assessment we combined WRIAs 3 and 4, the lower and upper Skagit River WRIAs, into a single WRIA.

Relative Conservation Value

Our relative conservation value index for AUs was based on expert judgment. Judgments were made regarding the components of the index, how to assemble them, and the relative influence of each component. The rationale for these judgments is described by our conceptual model of conservation value which is presented in Part 3 of this report. The components of the index are organized as 5 tiers (Figure B1). The bottom tier has three components: the density of wetlands and undeveloped floodplains, local salmonid habitats, and the accumulative downstream habitats. Local habitats are the salmonid habitats inside an AU, and accumulative downstream habitats are the salmonid habitats outside and affected by an AU.

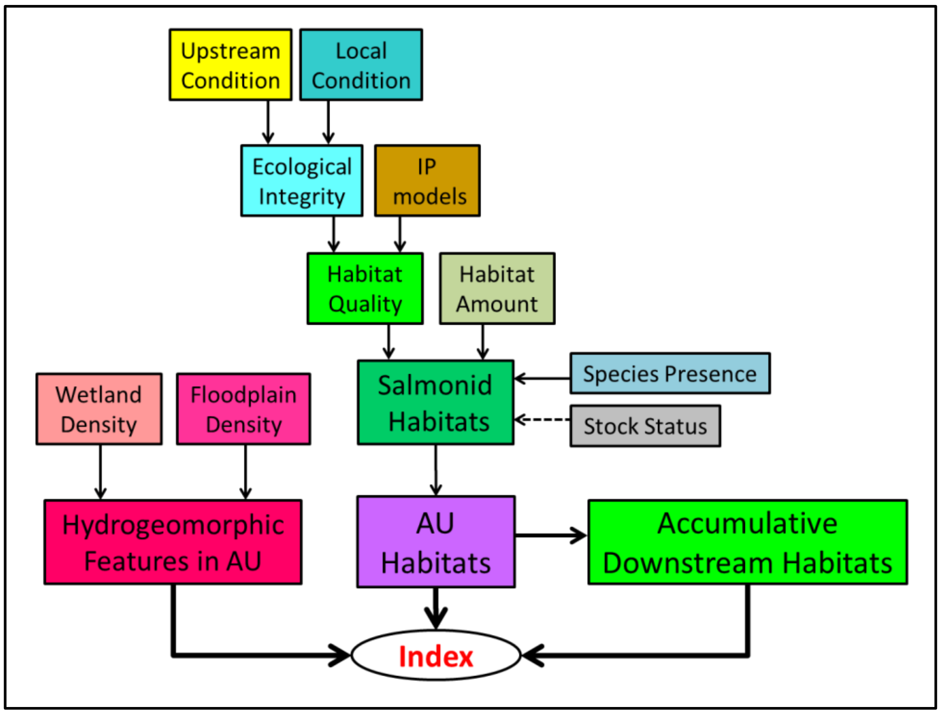


Figure B1. Components of the relative conservation value index. The three main components are hydrogeomorphic features in the AU, salmonid habitats in the AU, and accumulative downstream salmonid habitats. The dotted line from stock status indicates that this component was optional and included only as a secondary analysis done to examine the effects of stock status on relative conservation value. IP = intrinsic potential.

Hydrogeomorphic Features

Hydrogeomorphic features – i.e., wetlands and undeveloped floodplains – are included in the assessment because they are crucial to maintaining the quality of salmonid habitats. Spatial data for wetlands was obtained from Department of Ecology (Stanley et al. 2011). We refined the wetland data layer by overlaying it with a land cover/ land use data layer (C-CAP 2008) and removing wetlands co-incident with urban or agricultural land uses, i.e., the following C-CAP land use categories: low, medium, and high intensity developed, open space developed, pasture, and cultivated.

There were no readily available comprehensive spatial data for floodplains in the Puget Sound Basin. Hence, we contracted with M2 Environmental Services to generate a floodplain layer using digital elevation models (DEMs). We refined the floodplain layer with levee data from the Corps of Engineers

(2005). Lines depicting the locations of levees were overlaid onto the floodplain layer and floodplains were manually edited to remove areas behind levees. The area of functional floodplains was calculated by removing areas that were co-incident with “developed” land uses in C-CAP (low, medium, and high intensity developed, and open space developed).

With respect to hydrogeomorphic featues, an AU has high value if it has a high density of wetlands and undeveloped floodplains or contains a high proportion of a WRIA’s wetlands and undeveloped floodplains (Figure B2). Hence, relative value was calculated two ways: the percent of an AU’s area covered by hydrogeomorphic features, and the percent of a WRIA’s hydrogeomorphic feature area contained within an AU (Figure B3).

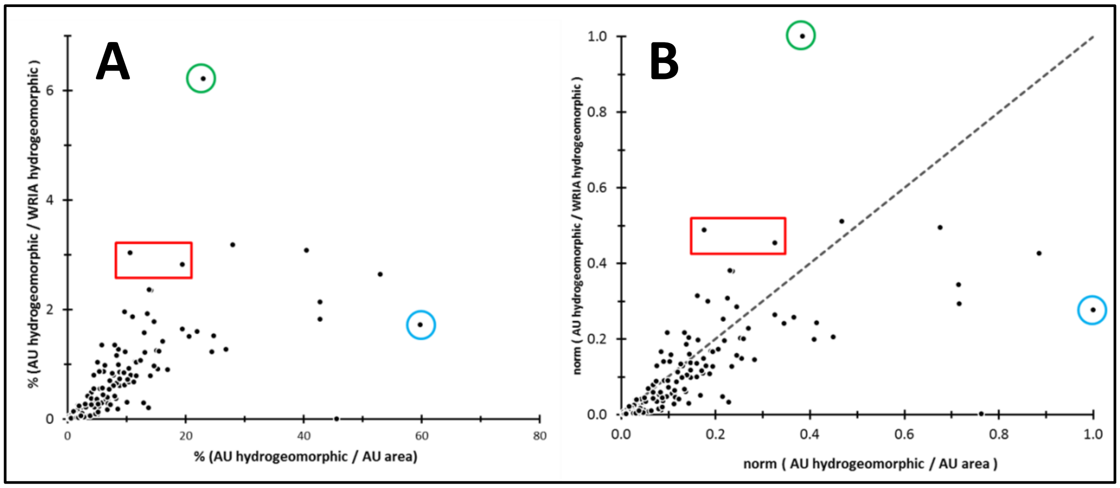


Figure B2. (A) With respect to hydrogeomorphic features, an AU has high value if it has a high density of features (blue circle) or contains a high proportion of a WRIA’s features (green circle). (B) Data in graph A normalized such that AUs in green and blue circles have the same relative value. Gray dashed line is 1:1 slope. Data from WRIA 8.

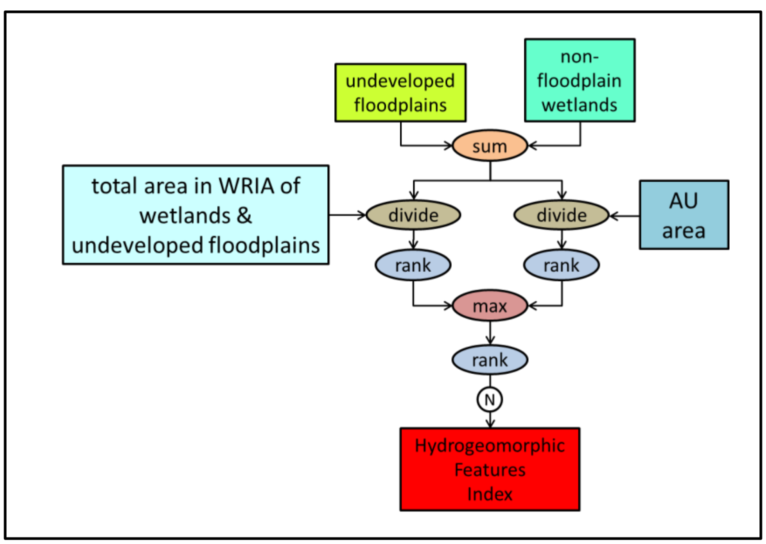


Figure B3. Process for calculating hydrogeomorphic features index. The ranks are normalized to range from 0 to 1.

AU Habitats

AU habitat value was a function of habitat quality, habitat amount, and presence category. WDFW's FishDist database (Figure B4) was the source of all spatial data on the presence of salmonids in rivers and streams. There are three categories of presence in FishDist: documented, presumed, and potential (Appendix C). "Documented" means the water body is known to be presently utilized by that fish species. "Presumed" means reliable documentation of fish use is lacking, but based on the available data and best professional opinion fish are thought to occur at that location. "Potential" means the water body meets the basic criteria for "presumed" but is unused by fish due to artificial obstructions, degraded habitat quality, or extinction of local fish populations. We also included occurrence data for transported (i.e., trap-and-haul) stocks. FishDist data for eight salmonid species and major life history variants – Chinook, coho, pink, chum, sockeye, kokanee, steelhead, rainbow trout, cutthroat, and bull trout – were transferred to reaches in the NetTrace channel network. For the purposes of this analysis, we equated presumed with documented but assigned lesser value to water bodies where a salmonid species had potential presence (Table B5). To simplify the analysis we lumped kokanee with sockeye, and where steelhead and rainbow trout co-occur we lumped them together also.

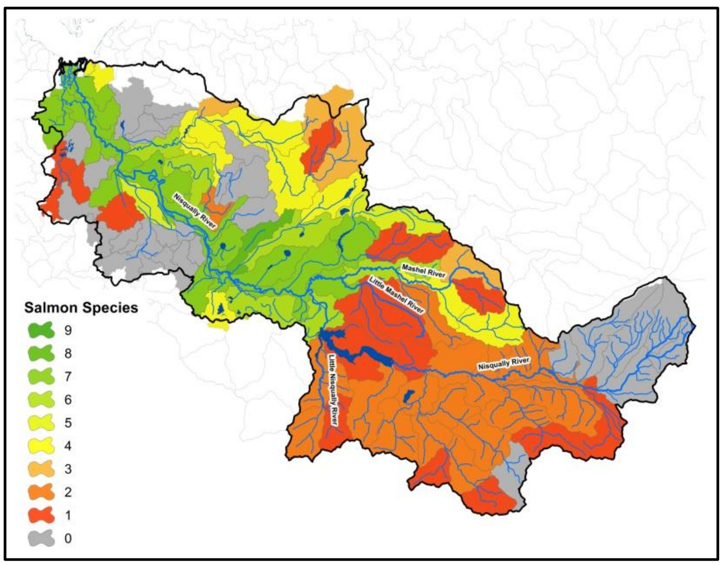


Figure B4. Salmonid species richness in the Nisqually Watershed (WRIA 11) according to WDFW's current data in the FishDist database. Categories denoting presence were limited to documented, presumed, and potential. Tables B3 and B4 give data summaries for each species and WRIA, respectively.

Habitat quality was the weighted geometric mean of intrinsic potential and ecological integrity (Figure B5). *Intrinsic potential* (IP) is an index unique to each salmonid species that indicates the potential habitat quality of a stream reach (Burnett et al. 2007). We currently have IP models for steelhead, coho, and Chinook (Figure B6). None of these models were developed for the Puget Sound Basin. The steelhead and coho models (Burnett et al. 2007) were developed for juvenile salmon in western Oregon, and both are a weighted geometric mean of the form:

$$IP = (I_{MF}^{a} I_{VW}^{b} I_{CG}^{c})^{\frac{1}{(a+b+c)}} \quad \text{(B1)}$$

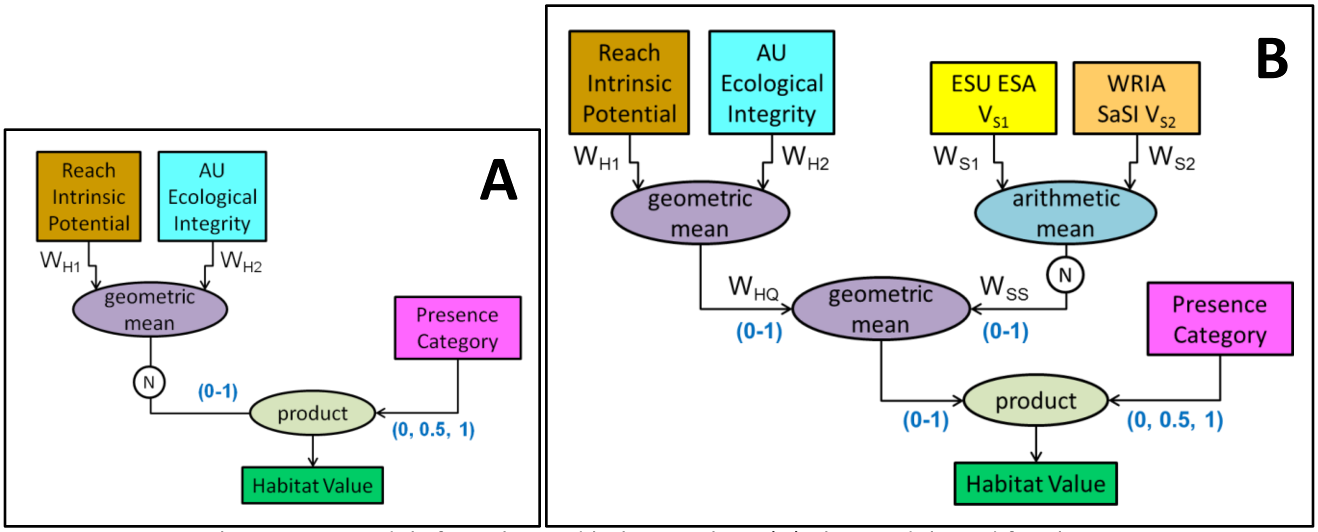


Figure B5. Two alternative models for salmonid habitat value. (A) The model used for this assessment did not include species status. (B) An alternative model that includes species status. Both models are for those salmonid species for which we have an intrinsic potential (IP) model. Values assigned to weights, Wx, given in Table B5. ESU ESA is the status of a salmonid species' evolutionary significant unit under the Endangered Species Act. WRIA SaSI, is the stock status according to the WDFW's Salmonid Stock Inventory; stocks are often confined to a WRIA. Values assigned to ESA and SaSI status shown in Table B6. For salmonid species that currently lack an IP model, intrinsic potential equals 1. N stands for normalization which is done within WRIAs.

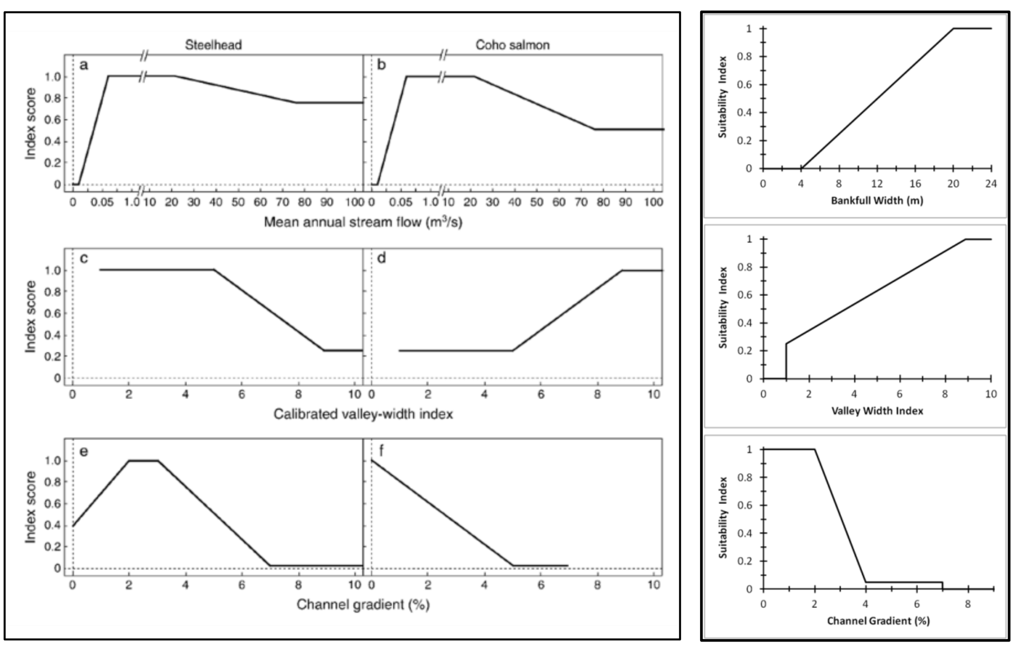


Figure B6. Left panel: Intrinsic potential models for juvenile steelhead and coho salmon from Burnett et al. (2007). Right panel: Intrinsic potential models for fall Chinook spawning habitat from Busch et al. (2011).

where a, b, and c are subjective weights that determine the relative influences of each factor; IMF is an index for mean annual flow; IVW is an index for valley width; and ICB is an index for channel gradient. Burnett et al. (2007) set a, b, and c to 1. Values for the independent variable values of the IP models were generated by NetTrace for every reach.

We used two IP models for Chinook: an IP model for fall Chinook spawning habitat in the lower Columbia River (Busch et al. 2011), and a model for spring/summer Chinook (i.e., stream-type Chinook) in the Interior Columbia Basin (Cooney and Holzer 2006). The IP relationships for Puget Sound Chinook are likely to be different than either of these models, but based on discussions with the models' creators (S. Busch and T. Cooney, pers. commun.) we believed that the models were adequate for our purposes. Namely, to make distinctions in relative conservation value at a coarse spatial scale. The steelhead model was also applied to rainbow trout. We did not have IP models for chum , pink or sockeye salmon, sea-run cutthroat, and bull trout. For those salmonid species that lack an IP model, intrinsic potential was set equal to 1, and consequently, habitat quality was only a function of ecological integrity.

Ecological integrity describes the degree to which an ecosystem is whole, intact, or undisturbed (Andreasen et al. 2001). Ecological integrity has been assessed and mapped using spatial data such as roads, land use, land cover, housing density, or human population density that served as surrogates for ecosystem degradation (Mattson and Angermeier 2007, Theobald et al. 2010, Esselman et al. 2011). To develop our index of aquatic ecological integrity we utilized two studies that found significant relationships between indices of biological integrity (IBIs) and the proportion of a watershed covered by certain land covers or land uses. DeGasperi et al. (2009) found significant linear regression relationships between a benthic macro-invertebrate IBI (B-IBI) devised for Puget Lowlands and the percent of a watershed that is impervious surface, and between the Puget Lowland B-IBI and the percent of a watershed that is forested. Mebane et al. (2003) found a significant linear regression relationship between a fish IBI devised for Pacific Northwest coldwater rivers and the percent of a watershed that is "disturbed" land.

Separately, DeGasperi et al (2009) and Mebane et al. (2003) provide incomplete descriptions of ecological integrity. Fortunately these studies are complementary in at least two ways. First, they cover different species. The IBI used by DeGasperi et al. (2009) is based on benthic macro-invertebrates only and the IBI of Mebane et al. (2003) is based on fish only, but combined they cover many of the aquatic animal species found in streams of the Puget Sound Basin. The two studies cover watersheds with different levels of disturbance. DeGasperi et al. (2009) covered highly disturbed watersheds − 90% of their sampling locations were located in watersheds with 32 to 96 % nonforest land cover. In contrast, Mebane et al (2003) covered watersheds with low levels of disturbance − 90% of their western Oregon sampling locations occurred in watersheds with 0 to 29 % disturbed land cover (Figure B7).
Both DeGasperi et al. (2009) and Mebane et al. (2003) performed straight line regressions on their data. Using data published in each article, we conducted are own analyses and found better fits to the data were obtained with power functions. A better fit to the data in Mebane et al. (2003) was obtained with a concave function. The steep concave shape is consistent with other studies finding similarly shaped relationships between fish abundance and forest harvesting (e.g., Ripley et al. 2005) and with expert opinion regarding the effects of road density on aquatic ecological integrity (Quigley et al. 2001). By definition, ecological integrity for a completely undisturbed watershed should be perfect, i.e., integrity equals 100. Hence, for the purpose of an index, another regression on the Mebane et al. (2003) data was done in which the y-intercept was fixed at 100. We then applied a weighted average to the two functions with the weights being 1-%disturbed/ 10.75 and %disturbed/10.75 for the functions with and without the fixed y-intercept, respectively, and where %disturbed equal to 10.75 is where the two

functions intersect. In other words, when %disturbed was approximately zero the function with the fixed y-intercept dominated the weighted average, when %disturbed was approximately 10.75 the function without the fixed y-intercept dominated, and for %disturbed greater than 10.75 the weighted average equaled the function without the fixed y-intercept. The resulting function we called the Mebane-derived integrity index (Figure B7).

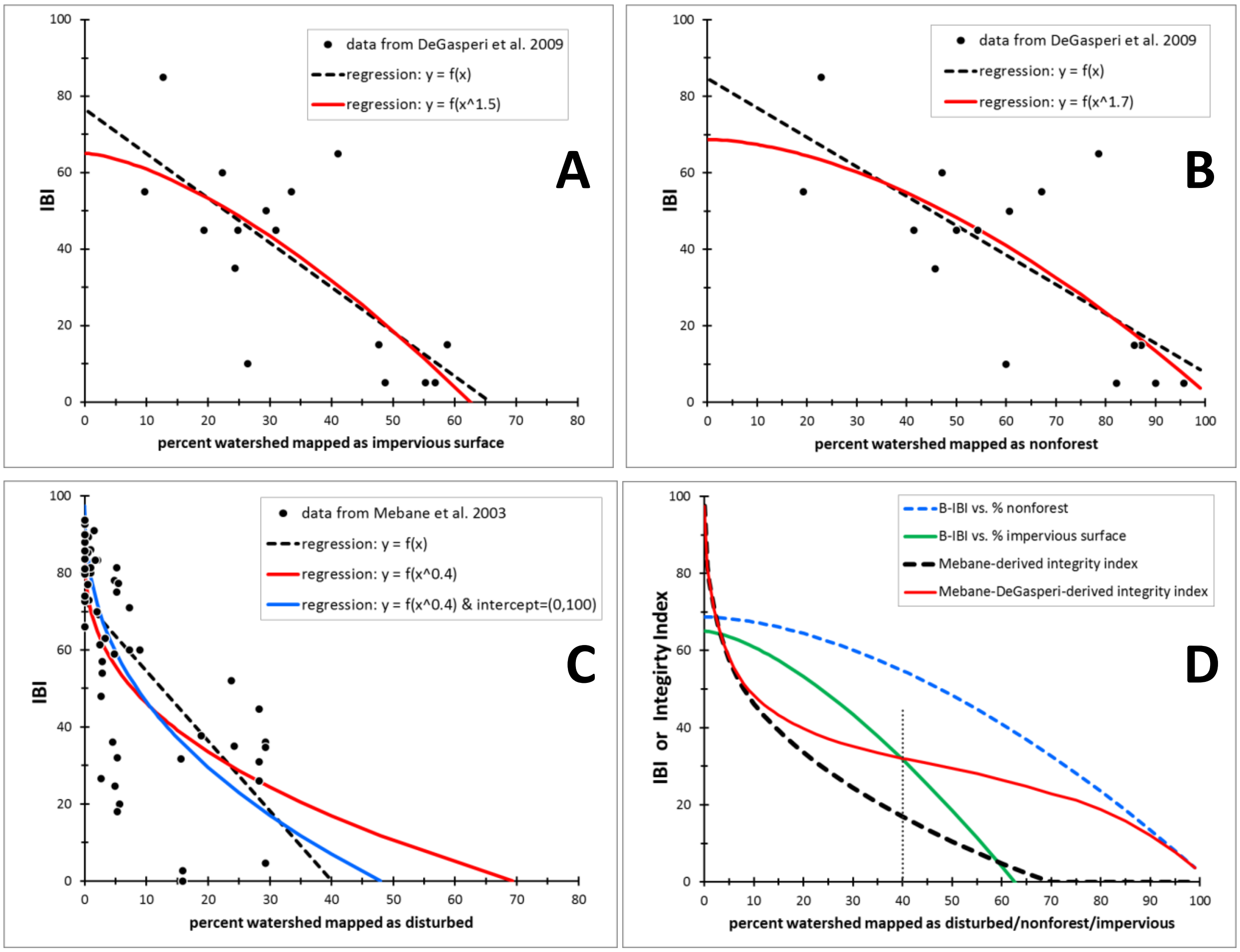


Figure B7. Empirical relationships used to derive the aquatic ecological integrity function. A: function with $x^{1.5}$ transform was slightly better fit to data (r^2=0.536, p <0.005) than straight line regression (r^2=0.535, p <0.005). B: function with $x^{1.7}$ transform was slightly better fit to data (r^2=0.529, p <0.005) than straight line regression (r^2=0.522, p <0.005). C: function with $x^{0.4}$ transform was better fit to data (r^2=0.554, p <0.0001) than straight line regression (r^2=0.447, p <0.0001). For purposes of an index, regression on data in Mebane et al. (2003) was forced through y-intercept of 100, and the two regressions were combined to create the Mebane-derived integrity index (shown in panel D). D: upper dashed curved and lower dashed curve were combined through a weighted average to form a Mebane-DeGaspari derived integrity index. The vertical dotted line demarks a change in the integrity function – to the left the function is always the aquatic ecological integrity index (red line); to the right the function is the smaller of either the Mebane-DeGasperi-derived ecological integrity index or the B-IBI vs. % impervious surface relationship (green line).

From the data in DeGasperi et al. (2009) we generated separate regression relationships for B-IBI versus percent impervious surface and for B-IBI versus percent nonforest (Figure B7). Better fits to the data were obtained with convex functions. We used both relationships in our calculation of ecological integrity.

We combined the Mebane-derived integrity index with the convex B-IBI versus percent nonforest relationship with a weighted average with the weights being 1-%nonforest/100 and %nonforest/100 for the Mebane-derived index and DeGasperi derived relationship, respectively. In other words, when %nonforest was approximately zero the Mebane-derived index dominated the weighted average and when %nonforest was approximately 100 the DeGasperi-derived relationship dominated. The resulting function we called the Mebane-DeGasperi-derived integrity index.

According to the relationships derived from the DeGasperi et al. (2009) data, the amount of impervious surface in a watershed has a much more severe impact on B-IBI than the amount of nonforest. To account for the impacts of impervious surface, we calculated two index values: one using the convex B-IBI relationship for percent impervious surface and one using the Mebane-DeGasperi-derived integrity index. We then selected the lesser of the two index values as our overall index of aquatic ecological integrity. See Box B1 for detailed description of function used to calculate the aquatic integrity index.

Our index of aquatic ecological integrity is ultimately based on land cover. We enhanced two existing land cover layers. First, for impervious surface, we started with the impervious surface layer of the 2006 National Land Cover Database (NLCD; Fry et al. 2011), which contains integer values from 0 to 100 that report the percent impervious surface within each raster cell. This layer consists of data collected via satellite which omit many narrow roads in forested areas (i.e., logging roads). To improve the accuracy of the NLCD layer we merged it with a rasterized roads layer (DNR 2011). Hence, a road traversing a raster cell equals approximately 19% impervious surface.

We assumed that logging roads are typically about 20 ft wide. Raster cells in the NLCD layer are 107.13 x 107.13 ft. The rasterized road and NLCD impervious layers were combined by comparing raster cells and taking the maximum value. This resulted in 5.9 percent of NLCD cells being changed to 19% impervious. The percent impervious surface in an AU was the average percent of all raster cells in the AU.

Second, for land cover, we started with the Coastal Change Analysis Program's 2006 land cover data layer (C-CAP 2008), which has 21 categories in the Puget Sound Basin. The C-CAP layer does not make distinctions between bare ground, grass, and scrub/shrub caused by human disturbance and those same categories resulting from of natural conditions, such as in wilderness areas. Using the Major Public Lands spatial data layer created by the Washington Department of Natural Resources we improved the thematic precision of C-CAP by making those distinctions. We assumed that bare ground, grass, and scrub/shrub in national parks, federal wilderness and roadless areas, state natural area preserves and natural resource conservation areas, and state city and county parks above 500 ft in elevation were the result of natural conditions, and these categories were reclassified as rock, natural grassland, and natural scrub/shrub, respectively. Furthermore, we reclassified some raster cells in the C-CAP layer using the ecological systems layer developed by the US Geological Survey's GAP Program (Sanborn 2007). Specifically, we extracted the prairie category from the ecological systems layer and substituted these prairie raster cells for the C-CAP cells. The main source for the prairie category was the 2005 oak woodland and grassland prairie data from the Washington Natural Heritage Program that was created from aerial photo interpretation and field surveys, and hence, is more accurate than C-CAP. The new category in C-CAP was called prairie & oak woodland. Our enhancements to the C-CAP layer resulted in a reclassification of 7.8% of raster cells.

In Mebane et al. (2003) disturbed land was defined as agricultural, residential, and urban, which we equated with the C-CAP land use categories high, medium, and low intensity developed, open space developed, pasture/hay, cultivated, bare ground, and grassland. The percent disturbed in an AU was the percent of the AU covered by these categories. We equated nonforest with the disturbed categories plus the scrub/shrub category.

The ecological integrity of aquatic habitats is governed by processes occurring both locally and remotely. Hence, we applied the ecological integrity functions to six zones that divided a drainage area along both lateral and longitudinal dimensions (Figure B8). The lateral zones were floodplains/riparian areas and uplands. The longitudinal zones were 1) the focal AU, 2) AUs immediately upstream of the focal AU, and 3) all other AUs in the upstream drainage area of the focal AU. Floodplains were generated by M2 Environmental Services using DEMs. Riparian areas were delineated using the Washington State Watercourse Hydrography (DNR 2006). Fish-bearing streams and shorelines of the state (F and S waters, respectively) were buffered by 200 ft, and non-fish-bearing and streams with no designation (N and X waters, respectively) were buffered by 150 ft. The floodplains/riparian areas were merged into one layer.

Ecological integrity index values calculated for these six zones were combined with a weighted arithmetic mean to yield a composite ecological integrity value for each AU. The equation was:

$$CEI = w_F \sum_{i=1}^{3} I_{Fi} P_{Fi} e^{-d(D_i - D_1)} \quad + \quad w_U \sum_{i=1}^{3} I_{Ui} P_{Ui} e^{-d(D_i - D_1)} \qquad \text{(B2)}$$

where i denotes sub-areas 1, 2, or 3 (see Figure B8); the subscripts F and U denote floodplain/riparian area and upland sub-divisions, respectively, of those sub-areas; I_{Fi} and I_{Ui} are the aquatic ecological integrity indices calculated for the six zones; P_{Fi} and P_{Ui} are the proportion of the AU's entire contributing drainage area covered by each zone; D_i is the mean distance of sub-areas 1, 2, or 3 from Puget Sound; w_F and w_U are subjective weights based on professional judgment that represent relative impact per acre of floodplains/riparian areas versus uplands (Table B5); and d is a constant set such that the impact per acre of zones 2 or 3 are attenuated by ½ when D_i-D_1 equals 5 miles.[20] I_{Fi} and I_{Ui} were calculated using the functions in Figure B7(D). M2 Environmental Services created a computer program that tabulates land cover categories in each of the 6 zones for every AU; that program was also used to derive D_i.

Habitat quality is the geometric mean of intrinsic potential and ecological integrity indices:

$$HQ = \left(IP^{W_{H1}}\ CEI^{W_{H2}}\right)^{\frac{1}{(W_{H1}+W_{H2})}} \qquad \text{(B3)}$$

where WH1 and WH2 are subjective weights determined through expert judgment (Table B4). Habitat quality is combined with species presence category (and optionally the species and stock status; Figure B5) to yield a reach's habitat value. Habitat value is calculated for each species present in a reach. Hence, up to eight values per reach must be combined into a single value. We derived two separate

[20] The value for d was 0.13863, which equals -ln(0.25)/10. Setting d to this value causes the integrity index of zones 2 or 3 to be multiplied by ¼ when D_i-D_1 equals 10 miles and multiplied by ⅛ when D_i-D_1 equals 15 miles.

reach-scale metrics that combine the eight habitat values (Figure B9): the maximum habitat value and the sum of all habitat units, which equal habitat value times the habitat amount (i.e., reach length). Habitat units are a convention used in habitat evaluation procedures (USFWS 1980). A reach contributes the largest amount of habitat units when it is a long and has high habitat value, but exceptionally long reaches with low habitat value and short reaches with exceptionally high habitat value can also contribute a large amount of habitat units.

Using only one of the two reach-scale metrics would fail to identify many high value reaches. Maximum habitat value identifies reaches that contain exceptionally high quality habitat for only one species, while the sum of habitat units identifies reaches with a large amount of habitat for many species. Hence, the reach habitats index (RHI) is the maximum of the two metrics (Figure B9). RHI is used in the calculation of accumulative downstream habitats.

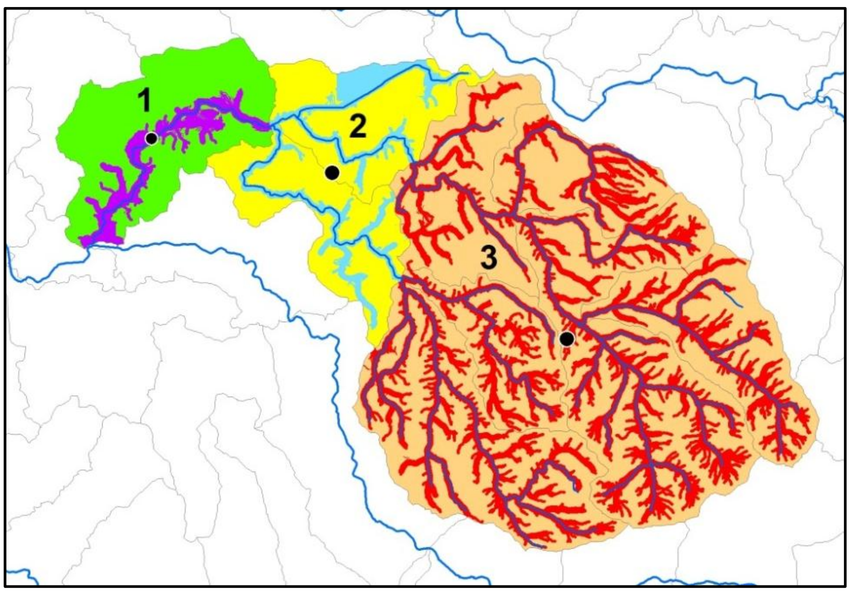


Figure B8. Six zones for which aquatic ecological integrity was calculated. The drainage area of each AU was divided into three sub-areas: 1) the focal AU, 2) AUs immediately upstream of the focal AU, and 3) all other upstream AUs. These sub-areas were further sub-divided into floodplains/riparian areas (purple, blue, and red) and uplands (green, yellow, and red). The index values (Figure B7) calculated for the six zones were combined through a weighted arithmetic average to yield the aquatic ecological integrity of the focal AU. Black dots represent the mean distance of each sub-area from Puget Sound. Gray lines are AU boundaries and blue lines are rivers and streams. Only rivers and streams mapped 1 1:24,000 are shown.

AUs are small watersheds. Hence, the reach-scale habitat values and habitat units must be combined to yield a watershed-scale index. There are several ways an AU can be highly valuable for the conservation of salmonid habitats – when an AU contains a large amount of exceptionally high quality habitat for only one species; contains large amounts of habitat for many species, regardless of habitat quality; contains some intermediate amounts of high quality habitats for some species, or contains large amounts of moderate quality habitats for some species, etc. The watershed habitats index (WHI) can rate all these situations as having relatively high conservation value. WHI for an AU equals the maximum of either the

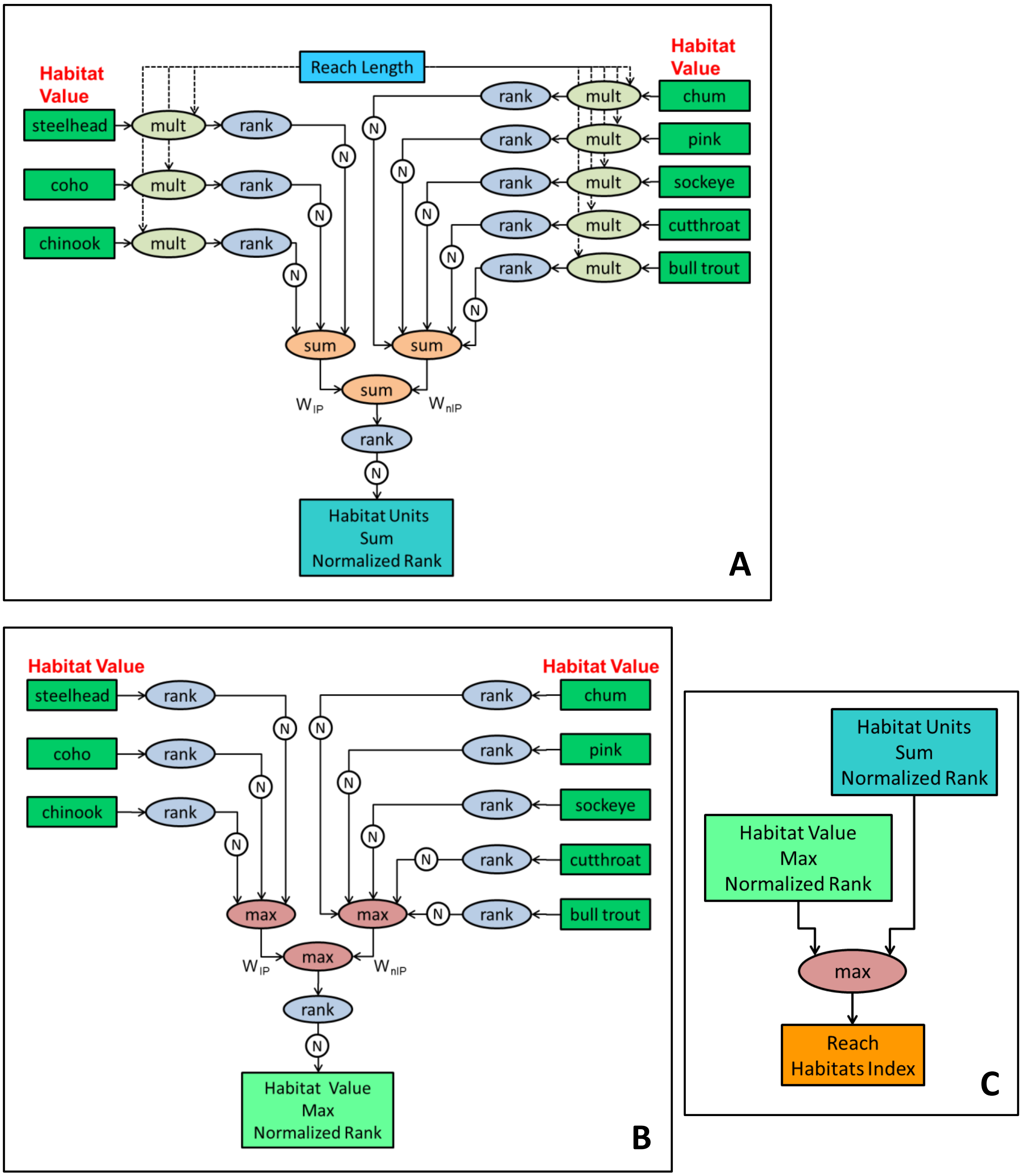


Figure B9. Calculation of reach habitats index (RHI). RHI combines two metrics: sum of habitat units (A), and maximum habitat value (B) which are calculated for each reach. In panels A and B, species on left have IP models and species on the right do not. Ranking and normalization are done within WRIAs. N stands for normalization which converts ranks to a range of values from 0 to 1. Values assigned to weights, W_x, in Table B6.

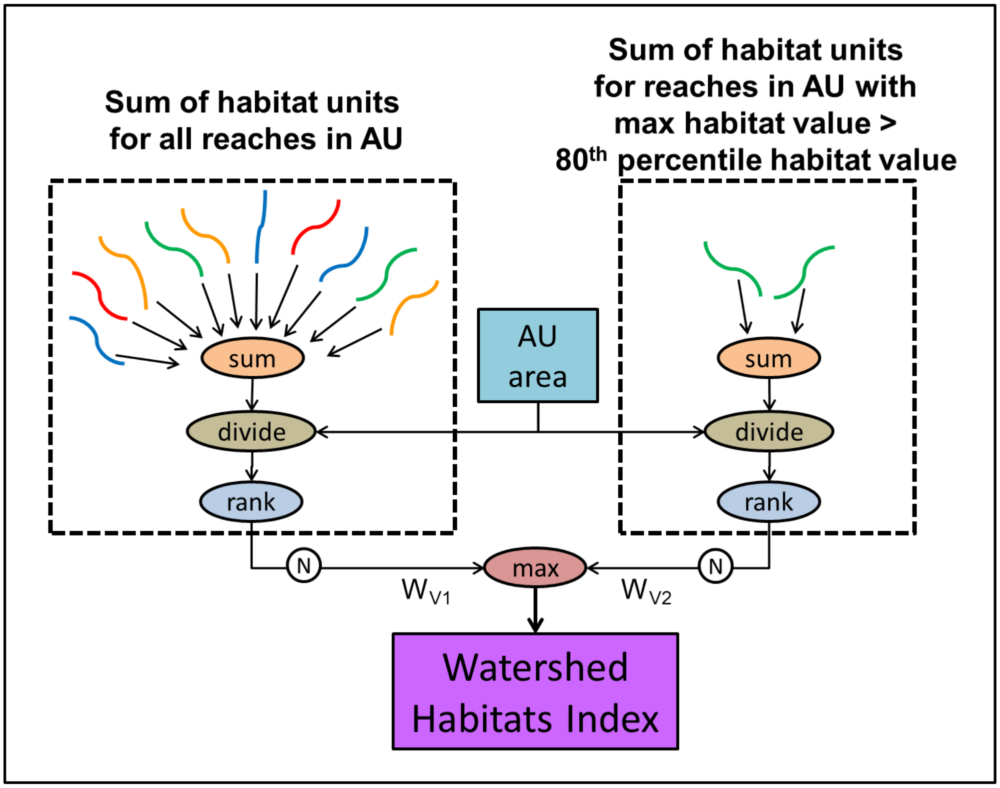


Figure B10. Watershed habitats index (WHI) is the maximum of either the sum of habitat units for all stream reaches in the AU or the sum of habitat units for all reaches in the AU that have maximum habitat value greater than the 80th percentile for the WRIA where the AU is located. Ranking and normalization are done within WRIAs. N stands for normalization which converts ranks to a range of values from 0 to 1. Values assigned to weights, W_x, in Table B5.

sum of habitat units for all stream reaches in the AU or the sum of habitat units for reaches in that AU with a maximum habitat value greater than the 80th percentile habitat value for the WRIA where the AU is located (Figure B10). In other words, WHI assigns a high value to AUs that either have a relatively large amount of habitat units or have a relatively large amount of high value habitat units. Before applying the maximum function, the two components of WHI were divided by AU area to yield a habitat unit density and normalized by their respective maximum values within the WRIA.

Accumulative Downstream Habitats

The downstream habitats component of an AU's relative conservation value was based on the habitat value (i.e., RHI) of all reaches downstream from that AU (Figure B11). The downstream calculations were performed for each reach in the AU, and the reach-level equation for accumulative downstream habitats was:

$$ADH_{ij} = \sum_{k=i+1}^{N} RHI_k \tag{B4}$$

where N is the number of reaches downstream of reach i in AU j, and RHI_k is the reach habitats index for downstream reach k. The AU-level ADH_j was the area-weighted mean ADH_{ij} for all reaches within AU_j, where the area was the drainage area adjacent to reach *I* in AU j. ADH was normalized by dividing all ADH_j by the maximum ADH_j in their respective WRIA.

M2 Environmental Services created a computer program that sums all downstream RHI_k for every stream reach and another computer program that calculated the drainage area adjacent to each reach.

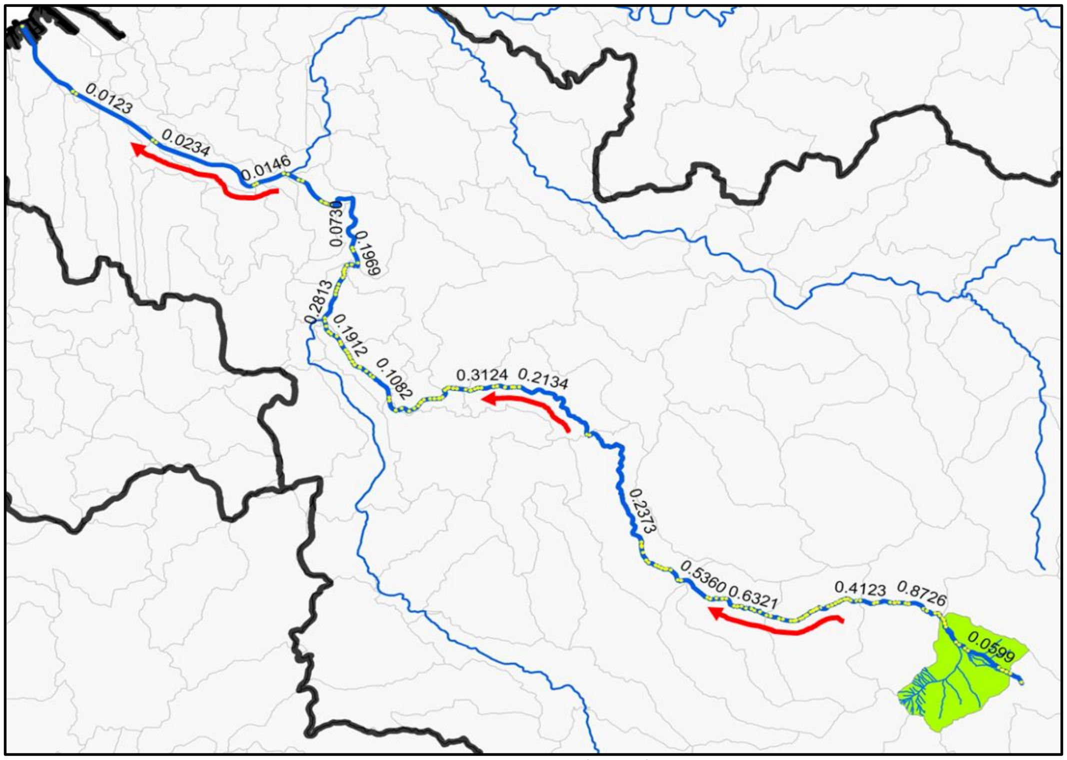


Figure B11. The accumulative downstream habitats (ADH) component of the index of relative conservation value. ADH for the green AU is the sum of RHI values downstream of the AU. Yellow dots mark breaks between adjacent stream reaches and numbers are hypothetical RHI values for each reach. Gray lines are AU boundaries, thick black lines are WRIA boundaries, and blue lines are rivers.

Relative Conservation Value Index

There are two basic perspectives on modeling the relative conservation value of places, and they reflect a quantity versus quality dichotomy. One perspective is that conservation value is best determined by a place's total contribution to habitat conservation, i.e., the quantity a place contributes. The other perspective is that value is best determined by a place's single most significant contribution, i.e., the quality a place contributes. These two perspectives can result in different rankings of places. For example, the former perspective would value a place with high species richness over a place with high species rarity, while the latter would value rarity over richness. Neither perspective should be ignored, so we examined conservation value both ways.

The perspective that favors quantity over quality can be informed by averaging the values of each AU for hydro-geomorphic features, local habitats (i.e., WHI), and accumulative downstream habitats (Figure B12). The perspective that favors quality over quantity can be informed by taking the maximum value of the three components. For the purposes of combining the three major components of relative conservation, their continuous values were ranked relative to other AUs in a WRIA and normalized to yield indices with a range of 0 to 1.

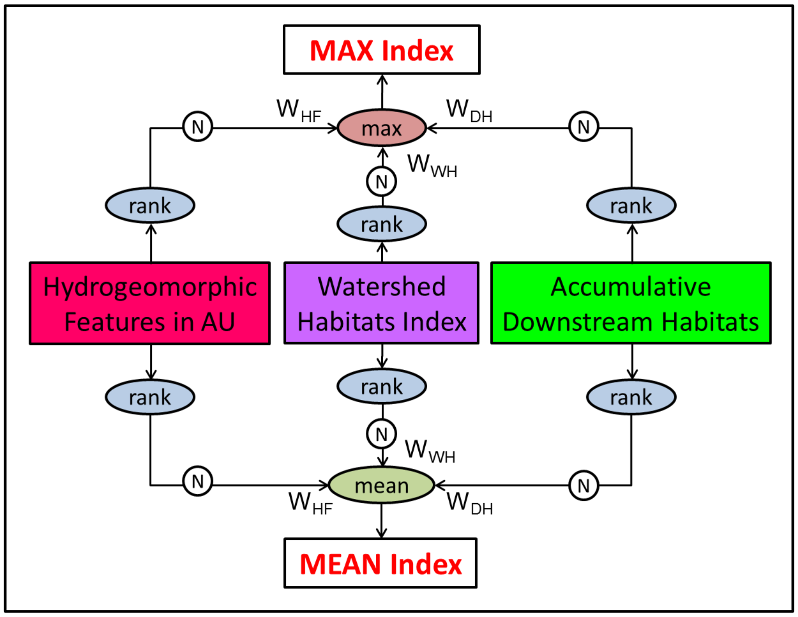


Figure B12. Two indices generated from the three components of relative conservation value. Ranking and normalization are done within WRIAs. N stands for normalization which converts ranks to a range of values from 0 to 1. W_x are weights, which were all set to 1 for this analysis.

Sensitivity Analysis

Sensitivity analysis was done by calculating WHI for all AUs, recalculating WHI after altering a single parameter by a small amount (e.g., 5%), and applying equations 1.2 and 1.3 to each AU. The process was repeated for parameters indicated in Table B5 and those in Table B6. The index value of each AU is effectively a separate model output, and hence each AU has its own sensitivity to each parameter. A mean sensitivity was calculated for each parameter by averaging over the separate sensitivities of all AUs.

Another sensitivity analysis examined how WHI changed in response to changes in model structure. We examined four major changes to the model: removing ecological integrity, removing intrinsic potential models, removing both ecological integrity and intrinsic potential (i.e., habitat quality), and including species status.

Species status could be another factor affecting the conservation value of salmonid habitats, however, because how species status is incorporated into the assessment entails a policy decision, we did not include status as a factor affecting relative conservation value. However, we were curious as to how species status would affect the results. The stock status index was the weighted arithmetic average of a stock's status under the Endangered Species Act (ESA) and its status according to the Salmonid Stock Inventory (SaSI). WDFW's SaSI database contains the stock status according to WDFW and WWTIT (2002) and the ESA status assigned by the National Marine Fisheries Service (Appendix C). At present in the Puget Sound Basin, Chinook, steelhead, Hood Canal summer chum, and bull trout are federally listed as threatened with extinction and coho are candidates for listing. SaSI data for 8 of the 10 salmonid species (SaSI does not include kokanee or rainbow trout) were transferred to reaches in the NetTrace channel network. Where the SaSI status was unknown or unassigned we assigned the average SaSI status for rated stocks of same species in Puget Sound Basin.

Uncertainty Analysis

Uncertainty analysis indicates the degree of confidence we should have in the model (see Part 1). Uncertainty analysis was done by assigning a uniform probability distribution to each of 11 parameters (Table B4). The distributions spanned the range of reasonable values for each parameter. Over 1,200 iterations, a parameter value was randomly selected from each distribution and the index was calculated for each AU. Only 1,200 iterations were performed because the large number of stream reaches in the Puget Sound Basin (> 700,000) results in a computationally intensive analysis. The index value of each AU is effectively a separate model output, and hence each AU has its own distribution of index values. The uncertainty associated with each AUs index value was calculated and depicted as a 90% confidence interval.

Box B1. Pseudo-code describing the algorithm used to calculate aquatic integrity index. Actual code was implemented with VisualBasic for Applications (VBA).

```
'function derived from DeGasperi et al. (2009) data using linear regression;  used when impervious surface > 40% of zone
If %imperv > 40 Then
      imperv_func = 65.0526  -  0.13139*%imperv ^ 1.5                  'slightly convex power function, see Figure B5
      imperv_index = max(imperv_func, 0)                               'Integ_Index cannot be less than zero
Else
      imperv_index = 100             'if %imperv <= 40, then imperv_index not used; so set imperv_index to large number
End

'function derived from Mebane et al. (2003) data using linear regression
disturb_funcA = 85.61359  -  15.7055*%disturb ^ 0.4                    'concave power function, see Figure B5
disturb_funcB = 100  -  21.2686*%disturb ^ 0.4                         'force function A through y-intercept = (100,0)
If disturb < 10.7543 Then                                              'functions A and B intersect at 10.7543
      disturb_weight = 1 - %disturb / 10.7543                          'weight is a function of distance from intersection
      disturb_func = disturb_funcA*(1 - disturb_weight) + disturb_funcB*disturb_weight        'average the functions
Else
      disturb_func = disturb_funcA     'if %disturb > 10.7543, then don't average functions, use function without forced y-intercept
End

'function derived from DeGasperi et al. (2009) data using linear regression
nonforest_func = 68.7142  -  0.02632*%nonforest ^ 1.7           'convex power function, see figure B5
integ_weight = 1 -% nonforest / 100                             'by definition, %nonforest will always be greater than %disturb

'average the two functions;  as % nonforest increases, less weight on disturb_func
combined_index = nonforest_func*(1 - integ_weight)   +   disturb_func*integ_weight

Integ_Index = min(imperv_index, combined_index)                 'the aquatic integrity index
```

Table B3. Summary of WDFW FishDist database by salmonid species. See Appendix C for definitions of presence categories. Units are miles.

	FishDist Presence Categories			documented + presumed		documented + presumed + potential	
species	documented	presumed	potential	miles	% FishDist Miles	miles	%FishDist Miles
Chinook	2345.2	169.5	185.9	2514.7	29.9	2700.6	32.1
Coho	4122.4	406.6	257.4	4529.0	53.9	4786.4	56.9
Chum	1845.4	184.5	148.7	2029.9	24.1	2178.6	25.9
Pink	1097.4	109.1	50.3	1206.5	14.3	1256.8	14.9
Sockeye	931.9	25.0	10.9	956.9	11.4	967.8	11.5
Kokanee	216.1	12.9	2.5	229.0	2.7	231.4	2.8
Steelhead	3871.4	552.7	287.4	4424.2	52.6	4711.6	56.0
Rainbow	1427.8	69.1	0.0	1496.9	17.8	1496.9	17.8
Coast Resident Cutthroat	5050.1	1144.0	47.7	6194.1	73.7	6241.8	74.2
Bull Trout	1696.0	496.1	7.9	2192.1	26.1	2200.0	26.2
total	22603.7	3169.5	998.6	25773.2		26771.8	

Table B4. Summary of WDFW FishDist database by WRIA. See Appendix C for definitions of presence categories. Units are miles.

	FishDist Presence Categories			documented + presumed		documented + presumed + potential	
WRIA	documented	presumed	potential	Mile	% FishDist Miles	Miles	% FishDist Miles
1	3004.8	509.2	78.7	3514.0	358	3592.7	366
2	12.9	5.3	0.7	18.2	126	18.8	131
3	1689.8	288.6	18.9	1978.5	390	1997.4	394
4	2931.8	232.7	41.8	3164.6	359	3206.4	364
5	1798.5	445.1	6.9	2243.7	377	2250.6	378
6	32.2	2.7	53.1	34.9	85	88.0	214
7	3850.9	227.7	17.0	4078.5	320	4095.5	321
8	1194.5	5.1	0.0	1199.6	311	1199.6	311
9	1053.3	70.2	198.6	1123.5	292	1322.0	343
10	1107.3	149.4	83.3	1256.7	289	1340.0	308
11	902.8	133.2	28.2	1036.0	305	1064.2	313
12	100.2	37.1	6.9	137.3	226	144.1	237
13	261.5	6.9	0.3	268.3	158	268.6	158
14	414.2	404.0	9.7	818.2	216	827.9	218
15	1255.4	178.6	16.8	1434.0	255	1450.7	258
16	1048.2	136.5	56.7	1184.7	278	1241.4	292
17	385.8	177.7	33.1	563.4	182	596.6	193
18	857.2	136.0	345.3	993.2	256	1338.6	344
19	702.4	23.6	2.5	726.0	268	728.5	269
total	22603.7	3169.5	998.6	25773.2	307	26771.8	318

Table B5. Subjective weights used in calculation of conservation value index. X indicates the parameter was examined through sensitivity analysis.

Symbol	Value	Name	Equation or Schematic	Sensitivity & Uncertainty Analyses
W_{HF}	1	hydrogeomorphic features	Figure B12	
W_{LH}	1	local habitats		
W_{DH}	1	accumulated downstream habitats		
W_{H1}	1	intrinsic potential	Figure B5	X
W_{H2}	1	aquatic ecological integrity	Equation B2	X
W_{Fi}	10	floodplains and riparian areas	Equation B3	X
W_{Ui}	1	uplands		X
d	0.1386	rate of attenuation		X
W_{HQ}	4	salmonid habitat quality	Figure B5	X
W_{SS}	3 or 0	salmonid status*		X
W_{S1}	3	Endangered Species Act status	Figure B5	
W_{S2}	1	Salmonid Stock Inventory status	(optional)	
W_{nIP}	1	species without IP models	Figure B9	X
W_{IP}	1	species with IP models		X
W_{V1}	1	sum of sum habitat units	Figure B10	
W_{V2}	1	sum of max habitat value		

* W_{ss} equals 1 when status is included in index and equals 0 when it is not included.

Table B6. Translation of FishDist presence categories and SaSI and ESA status to numeric values representing relative conservation value (see Appendix C). SaSI and ESA status used only in a secondary analysis. X indicates the parameter was examined through sensitivity analysis.

Status	Value	Sensitivity & Uncertainty Analyses
FishDist Presence Categories		
Documented	2	X
Presumed	2	X
Potential	1	X
Salmon and Steelhead Inventory Status, V_{S2}		
Critical	3	
Depressed	2	
Healthy	1	
Unknown	average for rated stocks of same species	
Unassigned		
Extinct	0	
Endangered Species Act Status, V_{S1}		
Endangered	3	
Threatened	2	
Candidate	1	
no designation	0	

Appendix C: Definitions of fish presence categories and species status categories

Presence categories in WDFW's FishDist database

Documented - Aquatic habitat that is documented to be presently utilized by fish (based on reliable published sources, survey notes, first-hand sightings, etc.). This includes habitat used by any life history stage for any length of time. This designation is applied to all stream sections downstream of a documented sighting to the next "Documented" habitat section (or to marine waters), unless otherwise indicated by a formal review group.

Presumed - Aquatic habitat lacking reliable documentation of fish use where, based on the available data and best biological opinion/consensus, fish are presumed to occur. For migratory fish, such habitat will extend upstream to the end of the stream OR to the first known natural barrier (including sustained 12% stream gradient or small stream size). Best biological judgment includes consideration of suitable (species-specific) habitat availability, life history strategies, proximity and connectivity to adjacent "Documented" habitat sections or logical extrapolation of range from similar systems.

Potential - Aquatic habitat that meets the basic criteria for "Presumed" but is unused by fish due to artificial (man-made) obstructions, degraded habitat quality, or extirpation of local fish populations. This category is used in cases where habitat could be made available to fish through removal of obstructions, improvement of habitat, or re-introductions of fish.

Transported, documented - Refers to non-hatchery stocks that are wholly supported by "trap and haul" operations to habitat that is otherwise inaccessible due to a man-made obstruction, and aquatic habitat is documented to be presently utilized by fish.[21]

Transported, presumed - Refers to non-hatchery stocks that are wholly supported by "trap and haul" operations to habitat that is otherwise inaccessible due to a man-made obstruction, and aquatic habitat lacks reliable documentation of fish use where, based on the available data and best biological opinion/consensus, fish are presumed to occur.[23]

Federal Endangered Species Act

Endangered - any species which is in danger of extinction throughout all or a significant portion of its range

Threatened - any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range

Candidate - species for which the National Marine Fisheries Service or U.S. Fish and Wildlife Service has sufficient information on their biological status and threats to propose them as endangered or threatened under the Endangered Species Act, but for which development of a proposed listing regulation is precluded by other higher priority listing activities

[21] Transported fish (i.e., trap and haul) were included in the analysis as either documented or presumed presence.

WDFW's Salmonid Stock Inventory

Stock - The fish spawning in a particular lake or stream(s) at a particular season, which fish to a substantial degree do not interbreed with any group spawning in a different place, or in the same place at a different season.

Critical - a stock of fish experiencing production levels that are so low that permanent damage to the stock is likely or has already occurred.

Depressed - a stock of fish whose production is below expected levels based on natural variations in survival levels, but above the level where permanent damage to the stock is likely.

Healthy - a stock of fish experiencing production levels consistent with its available habitat and within the natural variations in survival for the stock.

Appendix D: Miscellaneous Tables for Marine Shoreline Habitats Assessment

Table D1. Datasets not used in the marine shoreline habitats assessment.

Taxon	PHS	Reason for excluding	Units	Source
giant and bull kelp beds	Y	does not cover entire Puget Sound and redundant with DNR shorezone	Square feet	WDFW
harlequin duck concentrations	Y	high rate of false negatives in nearshore, and redundant with PSAMP survey data	Square feet	WDFW WSDM
kokanee salmon	Y	resident freshwater subspecies	count	WDFW Fishdist
rainbow trout	Y			
oyster		redundant with WDFW oyster data	Categorical: 0 = Absent 1 = 0-50% 2 = 50-100%	DNR Shorezone

Table D2. Bird species for which PSAMP survey data were included in assessment.

Common Name		
American Coot	Dunlin	Pigeon Guillemot
Ancient Murrelet	Eared Grebe	Pomarine Jaeger
Bald Eagle	Gadwall	Ring-Billed Gull
Barrows Goldeneye	Great Blue Heron	Red-Breasted Merganser
Black-Bellied Plover	Greater Yellowlegs	Rhinoceros Auklet
Belted Kingfisher	Green-Winged Teal	Red-Necked Grebe
Black Brant	Harlequin Duck	Red-Throated Loon
Black Oystercatcher	Herring Gull	Ruddy Duck
Black Scoter	Horned Grebe	Ruddy Turnstone
Black Turnstone	Hooded Merganser	Sabines Gull
Bonapartes Gull	Heermann's Gull	Sanderling
Brandts Cormorant	Killdeer	Snow Goose
Brown Pelican	Godwit	Surfbird
Blue-Winged Teal	Marbled Murrelet	Surf Scoter
California Gull	Mew Gull	Thayers Gull
Canvasback	Northern Pintail	Trumpeter Swan
Caspian Tern	Oldsquaw	Tufted Puffin
Common Goldeneye	Osprey	Western Grebe
Common Loon	Pacific Loon	Western Gull
Common Merganser	Pied-Billed Grebe	Western X Glaucous Winged Gull
Common Murre	Pelagic Cormorant	White-Winged Scoter
Double-Crested Cormorant	Peregrine Falcon	

Table D3. Unidentified bird taxa for which PSAMP survey data were included in assessment.

Unidentified Taxa Group	
Unidentifed Brachyramphus Murrelet	Unidentified Murre
Unidentified Albatross	Unidentified Murrelet
Unidentified Alcid	Unidentified Passerines
Unidentified Black-Wing Tip Gull	Unidentified Phalarope
Unidentified Cormorant	Unidentified Sandpiper
Unidentified Diving Duck	Unidentified Scaup
Unidentified Duck	Unidentified Scoter
Unidentified Goldeneye	Unidentified Sea Bird
Unidentified Grebe	Unidentified Small Alcid
Unidentified Large Grebe	Unidentified Small Gull
Unidentified Large Gull	Unidentified Small Shorebirds
Unidentified Large Shorebirds	Unidentified Swan
Unidentified Loon	Unidentified Teal
Unidentified Medium Shorebirds	Unidentified Tern
Unidentified Merganser	Unidentified Turnstone
	Unidentified Yellowlegs

Table D4. Birds in PSAMP survey data that were considered "at-risk" for purposes of this assessment.

Common Name	Scientific Name	Status
marbled murrelet	*Brachyramphus marmoratus*	federally threatened
brown pelican	*Pelecanus occidentalis*	state endangered
peregrine falcon	*Falco peregrinus*	state sensitive
bald eagle	*Haliaeetus leucocephalus*	state sensitive
common loon	*Gavia immer*	state sensitive
common murre	*Uria aalge*	state candidate
western grebe	*Aechmophorus occidentalis*	state candidate
Brandt's cormorant	*Phalacrocorax penicillatus*	state candidate
great blue heron	*Ardea herodias*	state monitor

Table D5. Amount of each shoreform (Shipman 2008) along the shorelines of Puget Sound. Shorezone types refers to number of shorezone types (Berry et al. 2001b) spatially intersecting each shoreform.

	shoreforms											
			embayment				beach			rocky coast		
	artificial	delta	barrier estuary	barrier lagoon	closed lagoon marsh	open coastal inlet	bluff-backed beach	barrier beach	pocket beach	plunging rocky shore	rocky platform /ramp	total
miles	234.9	192.3	101.6	38.3	3.1	152.7	950.5	273.5	86.3	115.9	316.6	2466
percent	9.5	7.8	4.1	1.6	0.1	6.2	38.5	11.1	3.5	4.7	12.8	100
shorezone types	21	14	10	8	3	17	27	18	27	25	32	202

Table D6. Percent of Puget Sound shorelines mapped as each shoreform-shorezone combination. Shoreforms from Shipman (2008) and shorezone types are those mapped by Berry et al. (2001b) and use the habitat types of Dethier (1990). Habitat types in gray do not exist in Puget Sound.

	Shoreforms											
			embayment				beach			rocky coast		
Habitat Types	artificial	delta	barrier estuary	barrier lagoon	closed lagoon marsh	open coastal inlet	bluff-backed beach	barrier beach	pocket beach	plunging rocky shore	rocky platform/ramp	total
bedrock, exposed							0.03		0.04	0.43	1.29	1.79
bedrock, partially exposed	0.03	0.01					0.32	0.01	0.17	0.92	3.16	4.64
bedrock, semi-protected	0.02	0.02				0.00	0.02		0.10	1.31	1.67	3.15
bedrock, protected						0.03			0.13	1.12	1.72	3.00
boulder, exposed												
boulder, partially exposed	0.01					0.01	0.11		0.00	0.20	0.45	0.78
boulder, semi-protected									0.00	0.02	0.03	0.06
boulder, protected							0.01		0.01	0.07	0.11	0.19
hardpan, exposed												
hardpan, partially exposed	0.01		0.01	0.03		0.02	1.50	0.18	0.00			1.76
hardpan, semi-protected						0.03	0.03					0.06
hardpan, protected												
cobble, exposed												
cobble, partially exposed									0.01		0.18	0.20
cobble, semi -protected												
cobble, protected												
mixed coarse, exposed							0.02	0.02				0.04
mixed coarse, partially exposed	0.75	0.06	0.11	0.05	0.07	0.11	14.20	3.52	0.32	0.10	1.49	20.79
mixed coarse, semi-protected	0.09	0.02	0.18	0.01		0.06	0.55	0.21	0.34	0.06	0.35	1.85
mixed coarse, protected	0.02		0.04			0.03	0.50	0.20	0.42	0.16	0.64	2.00

Habitat Types	artificial	delta	barrier estuary	barrier lagoon	closed lagoon marsh	open coastal inlet	bluff-backed beach	barrier beach	pocket beach	plunging rocky shore	rocky platform/ramp	total
gravel, exposed							0.24	0.04			0.00	0.29
gravel, partially exposed		0.11	0.07			0.00	1.37	0.54	0.14	0.02	0.19	2.45
gravel, semi -protected	0.01			0.04		0.03	0.05	0.01	0.07	0.01	0.03	0.25
gravel, protected	0.01					0.02			0.04	0.01	0.01	0.09
sand, exposed				0.00			0.02	0.02	0.00	0.00	0.00	0.04
sand, partially exposed	0.64	0.16	0.35	0.07		0.18	9.38	2.95	0.24	0.01	0.36	14.33
sand, semi -protected	0.02	0.14	0.23	0.15		0.15	0.40	0.26	0.11	0.00	0.07	1.54
sand, protected	0.00	0.19	0.08	0.00		0.01	0.04	0.04	0.16	0.00	0.07	0.60
mixed fine, exposed		0.37					0.00					0.37
mixed fine, partially exposed	0.41	0.52	0.55	0.13	0.03	1.37	7.35	2.05	0.32	0.09	0.47	13.30
mixed fine, semi -protected	0.26	0.16	0.82	0.52		1.40	0.30	0.15	0.16	0.00	0.06	3.85
mixed fine, protected	0.01			0.04		0.13	0.35	0.29	0.68	0.13	0.33	1.96
mud, exposed			0.02									0.02
mud, partially exposed	0.75	0.88	0.31	0.07		1.02	1.17	0.33		0.00	0.00	4.53
mud, semi-protected	0.28	0.23	0.88	0.12		1.16	0.18	0.04	0.00	0.01	0.04	2.94
mud, protected				0.01		0.03		0.02	0.01	0.01	0.01	0.09
organic, exposed		0.11				0.04						0.16
organic, partially exposed		1.50	0.03	0.04		0.00	0.01	0.00				1.59
organic, semi-protected	0.14		0.37	0.15	0.03	0.06		0.00			0.00	0.75
organic, protected		2.50										2.50
reef, exposed												
reef, partially exposed	0.08	0.27	0.03	0.04		0.06	0.05	0.05	0.00		0.01	0.61
reef, semi-protected												
reef, protected												
artificial, exposed	0.02											0.02
artificial, partially exposed	3.46	0.17	0.04	0.03		0.08	0.29	0.11	0.01		0.08	4.27
artificial, semi-protected	2.33	0.38	0.01	0.05		0.13	0.05			0.00	0.01	2.96
artificial, protected	0.16							0.01	0.00	0.01	0.01	0.19
total	9.53	7.80	4.12	1.55	0.13	6.19	38.54	11.09	3.50	4.70	12.85	100

Table D7. Relative likelihood of occurrence used in calculation of relative habitat value index. Blank cell means probability equals zero.

Shoreform	Habitat Type	habitat probability	Abalone	Clam , Hardshell	Clam, Subtidal	Dungeness Crab	Oyster	Shrimp	Urchin	Geoduck	Sand Lance	Surf Smelt
			Species (or species group)									
Artificial	bedrock, partially exposed	0.0006	0.167			0.667			0.833			
	bedrock, semi-protected	0.0003			0.667	0.667						
	boulder, partially exposed	0.0001										1.000
	hardpan, partially exposed	0.0004				0.250				0.250		
	mixed coarse, partially exposed	0.0107		0.138	0.037	0.239		0.037		0.229	0.138	0.220
	mixed coarse, semi-protected	0.0004							0.250			
	mixed coarse, protected	0.0001		1.000								
	gravel, semi -protected	0.0001				1.000	1.000			1.000		
	gravel, protected	0.0002										
	sand, partially exposed	0.0065		0.106		0.379	0.015	0.121		0.288	0.212	0.212
	sand, semi -protected	0.0006		0.500		0.167		0.333				
	sand, protected	0.0001										
	mixed fine, partially exposed	0.0073		0.162	0.068	0.324	0.351	0.054		0.297	0.203	0.311
	mixed fine, semi -protected	0.0037				0.105	0.026	0.026			0.026	0.053
	mixed fine, protected	0.0002									0.500	
	mud, partially exposed	0.0059		0.033		0.383	0.183				0.083	0.050
	mud, semi-protected	0.0034		0.171		0.029						
	organic, semi-protected	0.0010		0.100		0.100						
	reef, partially exposed	0.0007				0.857	0.143					
	artificial, exposed	0.0002			1.000			0.500	0.500			
	artificial, partially exposed	0.0222	0.004	0.097	0.031	0.434	0.027	0.022	0.018	0.044	0.053	0.102
	artificial, semi-protected	0.0122		0.032	0.016	0.185			0.016			0.024
	artificial, protected	0.0007			0.143	0.143		0.143				0.143
Barrier Beach	bedrock, partially exposed	0.0002	0.000	0.000	0.000	1.000	0.000	0.000	1.000	0.000	0.000	0.500
	hardpan, partially exposed	0.0024				0.125			0.000			0.167
	mixed coarse, exposed	0.0002		0.500		0.500	0.000			0.500	0.000	0.000
	mixed coarse, partially exposed	0.0441		0.220	0.089	0.332				0.283		
	mixed coarse, semi-protected	0.0018	0.000	0.333	0.333	0.000	0.000	0.222	0.611	0.056	0.111	0.056
	mixed coarse, protected	0.0017		0.235	0.176			0.471	0.412	0.059	0.176	0.000
	gravel, exposed	0.0004		0.750	0.500	0.750	0.000	0.000	0.000	0.750	0.250	
	gravel, partially exposed	0.0024		0.250	0.083	0.375				0.375	0.208	
	gravel, semi -protected	0.0001	1.000	0.000	0.000	0.000	0.000	0.000	1.000	0.000	0.000	0.000
	sand, exposed	0.0002	0.000						0.000			

	sand, partially exposed	0.0432		0.250		0.325				0.361	0.280	
	sand, semi -protected	0.0017		0.412	0.176	0.412	0.176	0.294	0.176	0.118	0.294	0.176
	sand, protected	0.0006		0.333	0.167	0.000	0.000	0.000	0.500	0.000	0.167	0.167
	mixed fine, partially exposed	0.0323	0.000	0.331	0.094				0.021		0.176	0.377
	mixed fine, semi -protected	0.0020		0.050	0.050	0.250	0.150	0.000	0.000	0.100	0.100	0.100
	mixed fine, protected	0.0041		0.167	0.000	0.381	0.000			0.000	0.048	0.048
	mud, partially exposed	0.0040		0.317		0.195		0.000	0.000		0.024	0.122
	mud, semi-protected	0.0008		0.250	0.000	0.125	0.000				0.125	0.375
	mud, protected	0.0002		1.000		0.500					0.000	0.000
	organic, partially exposed	0.0001		0.000		0.000						
	organic, semi-protected	0.0001										
	reef, partially exposed	0.0006		0.833								
	artificial, partially exposed	0.0017		0.412	0.059	0.647				0.118	0.353	0.176
	artificial, protected	0.0001		1.000	1.000	0.000	0.000	1.000	0.000	0.000	0.000	0.000
Barrier Estuary	hardpan, partially exposed	0.0003		0.000	0.000			0.000				
	mixed coarse, partially exposed	0.0010	0.000	0.200	0.000	0.600	0.000	0.000	0.000	0.000	0.000	0.100
	mixed coarse, semi-protected	0.0005								0.000		
	mixed coarse, protected	0.0001		1.000		1.000						0.000
	gravel, partially exposed	0.0002				0.500	0.000					0.000
	sand, partially exposed	0.0030		0.161		0.161					0.129	
	sand, semi -protected	0.0016			0.000	0.188			0.125		0.000	
	sand, protected	0.0002		0.000		0.000	0.000		0.500		0.000	0.000
	mixed fine, partially exposed	0.0054		0.091		0.145	0.145		0.000			0.255
	mixed fine, semi -protected	0.0041	0.000						0.000			
	mud, exposed	0.0001		0.000		0.000	0.000			0.000	0.000	0.000
	mud, partially exposed	0.0014		0.071	0.000	0.286	0.000			0.000	0.143	0.214
	mud, semi-protected	0.0034										
	organic, partially exposed	0.0002		0.000		0.000					0.000	0.500
	organic, semi-protected	0.0013		0.000		0.077	0.077				0.154	
	reef, partially exposed	0.0004										0.000
	artificial, partially exposed	0.0002				0.000	0.000		0.000		0.000	0.500
	artificial, semi-protected	0.0001					0.000			0.000	0.000	
Barrier Lagoon	hardpan, partially exposed	0.0002										0.000
	mixed coarse, partially exposed	0.0007										
	mixed coarse, semi-protected	0.0001	0.000	0.000	0.000	0.000	0.000		0.000	0.000	0.000	0.000
	gravel, semi -protected	0.0001				0.000	1.000					
	sand, exposed	0.0001		0.000			0.000	0.000			1.000	0.000
	sand, partially exposed	0.0015										
	sand, semi -protected	0.0008					0.375					
	sand, protected	0.0001									0.000	
	mixed fine, partially exposed	0.0015		0.267		0.200	0.133					0.333
	mixed fine, semi -protected	0.0026		0.115		0.000	0.077				0.077	0.115
	mixed fine, protected	0.0002							0.500			

	mud, partially exposed	0.0005		0.400		0.200	0.000				0.000	0.200
	mud, semi-protected	0.0011		0.182			0.091				0.000	0.182
	mud, protected	0.0001							0.000		0.000	
	organic, partially exposed	0.0002		0.000		0.000						0.500
	organic, semi-protected	0.0008		0.000		0.000	0.125					0.000
	reef, partially exposed	0.0002				0.000						
	artificial, partially exposed	0.0001										0.000
	artificial, semi-protected	0.0002					0.000				0.000	
Bluff-backed Beach	bedrock, exposed	0.0004										0.000
	bedrock, partially exposed	0.0013										
	bedrock, semi-protected	0.0003										
	boulder, partially exposed	0.0006		0.000		0.000	0.000	0.000	0.500		0.000	0.000
	boulder, protected	0.0001							0.000			
	hardpan, partially exposed	0.0088			0.011	0.044			0.022			0.211
	hardpan, semi-protected	0.0002	0.000	0.500		0.000			0.000			
	mixed coarse, exposed	0.0005							0.000			
	mixed coarse, partially exposed	0.1071						0.110				
	mixed coarse, semi-protected	0.0028		0.071	0.214	0.107	0.000		0.429	0.071	0.000	0.107
	mixed coarse, protected	0.0032		0.121								
	gravel, exposed	0.0012								0.167		
	gravel, partially exposed	0.0047	0.188	0.271	0.229	0.354	0.167	0.188	0.396	0.292	0.250	0.458
	gravel, semi -protected	0.0002	0.500	0.000	1.000	0.000		0.000	1.000	0.000		0.000
	sand, exposed	0.0001		1.000	0.000	1.000		0.000	0.000	0.000		0.000
	sand, partially exposed	0.0751		0.212	0.045	0.292				0.453	0.203	
	sand, semi -protected	0.0032	0.061	0.242	0.061	0.303	0.030	0.091	0.121	0.091	0.061	0.121
	sand, protected	0.0005	0.400		0.200				0.600			
	mixed fine, exposed	0.0001		0.000		0.000						
	mixed fine, partially exposed	0.0604		0.335	0.127	0.493	0.494	0.202	0.005	0.351	0.187	0.387
	mixed fine, semi -protected	0.0045	0.000	0.174	0.130	0.152	0.022	0.087	0.043	0.000	0.043	0.130
	mixed fine, protected	0.0041	0.000	0.143	0.000				0.143			
	mud, partially exposed	0.0092										
	mud, semi-protected	0.0020		0.300	0.000	0.050	0.100	0.000	0.000	0.000	0.000	0.100
	organic, partially exposed	0.0001		0.000	0.000	0.000	0.000	0.000	0.000		0.000	0.000
	reef, partially exposed	0.0006		0.333		0.333		0.000	0.000		0.000	0.333
	artificial, partially exposed	0.0031		0.469	0.063	0.406	0.125		0.000	0.063	0.188	0.188
	artificial, semi-protected	0.0006		0.000		0.333	0.000					0.000
Closed Lagoon Marsh	mixed coarse, partially exposed	0.0002										
	mixed fine, partially exposed	0.0001		0.000		1.000	0.000					0.000
	organic, semi-protected	0.0001		0.000	0.000	0.000	0.000			0.000	0.000	0.000
Delta	bedrock, partially exposed	0.0003			0.000	0.000			0.000			
	bedrock, semi-protected	0.0005	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	mixed coarse, partially exposed	0.0008										
	mixed coarse, semi-	0.0002				0.500			0.000		0.000	

	protected										
	gravel, partially exposed	0.0005									
	sand, partially exposed	0.0008			0.500			0.000		0.000	
	sand, semi -protected	0.0009									
	sand, protected	0.0002									
	mixed fine, exposed	0.0003			0.667						0.000
	mixed fine, partially exposed	0.0022			0.364						
	mixed fine, semi -protected	0.0007						0.000	0.000		0.000
	mud, partially exposed	0.0043	0.000		0.295	0.000			0.000		
	mud, semi-protected	0.0010	0.100	0.000	0.200					0.000	0.000
	organic, exposed	0.0001	1.000	0.000	1.000						
	organic, partially exposed	0.0026			0.115						
	organic, protected	0.0060	0.000		0.033	0.000	0.000		0.000		
	reef, partially exposed	0.0009									
	artificial, partially exposed	0.0007			0.143						0.000
	artificial, semi-protected	0.0018	0.000		0.000						
Open Coastal Inlet	bedrock, semi-protected	0.0001	0.000		0.000		0.000				
	bedrock, protected	0.0008			0.000		0.000				
	boulder, partially exposed	0.0001			1.000						
	hardpan, partially exposed	0.0001	1.000	0.000	0.000	0.000					
	hardpan, semi-protected	0.0002			0.000	0.000					
	mixed coarse, partially exposed	0.0016									
	mixed coarse, semi-protected	0.0009	0.000	0.000	0.111	0.000	0.000	0.000	0.000	0.000	0.111
	mixed coarse, protected	0.0005									
	gravel, partially exposed	0.0001	0.000					0.000			
	gravel, semi -protected	0.0001			1.000						
	gravel, protected	0.0001	1.000								
	sand, partially exposed	0.0027	0.074								
	sand, semi -protected	0.0019	0.105		0.000			0.000			
	sand, protected	0.0001			0.000						0.000
	mixed fine, partially exposed	0.0078						0.013			
	mixed fine, semi -protected	0.0058									
	mixed fine, protected	0.0018	0.056		0.389						
	mud, partially exposed	0.0056	0.298								0.053
	mud, semi-protected	0.0079	0.275		0.050	0.088			0.000		0.025
	mud, protected	0.0005	0.000							0.000	0.000
	organic, exposed	0.0004						0.000			
	organic, partially exposed	0.0001	0.000		0.000	0.000		0.000	0.000	0.000	0.000
	organic, semi-protected	0.0004	0.500	0.000	0.000	0.000		0.000		0.000	0.000
	reef, partially exposed	0.0004	0.500		0.000			0.000			
	artificial, partially exposed	0.0011	0.364	0.000	0.000	0.091				0.000	0.182
	artificial, semi-protected	0.0014	0.000	0.000	0.000	0.000					0.000
Pocket Beach	bedrock, exposed	0.0019			0.105			0.316			
	bedrock, partially exposed	0.0065			0.364						
	bedrock, semi-protected	0.0073									

	bedrock, protected	0.0105		0.009								
	boulder, partially exposed	0.0001		0.000								
	boulder, semi-protected	0.0001		0.000			0.000					0.000
	boulder, protected	0.0004										
	hardpan, partially exposed	0.0001		0.000	0.000	0.000	0.000		1.000	0.000	0.000	0.000
	cobble, partially exposed	0.0004			0.000	0.000	0.000	0.000	1.000			
	mixed coarse, partially exposed	0.0098	0.080	0.030	0.030	0.260	0.010	0.100	0.230		0.010	0.090
	mixed coarse, semi-protected	0.0088	0.200			0.000		0.033	0.944			0.022
	mixed coarse, protected	0.0129	0.008	0.000		0.153		0.160	0.527			0.053
	gravel, partially exposed	0.0024							0.833			
	gravel, semi -protected	0.0026							0.885			
	gravel, protected	0.0013							0.308			
	sand, exposed	0.0001							0.000			
	sand, partially exposed	0.0048							0.224			
	sand, semi -protected	0.0028	0.429	0.000	0.000	0.286	0.000	0.000	0.786	0.000	0.000	0.107
	sand, protected	0.0032	0.030		0.030			0.000	0.303			0.061
	mixed fine, partially exposed	0.0103	0.010			0.524		0.067	0.067			0.114
	mixed fine, semi -protected	0.0027	0.185	0.037	0.000	0.111			0.481		0.000	0.111
	mixed fine, protected	0.0183	0.027						0.618			
	mud, semi-protected	0.0001	0.000						0.000			
	mud, protected	0.0003				0.667						
	reef, partially exposed	0.0002	0.000	0.000	0.000	0.000	0.000	0.000	1.000	0.000	0.000	0.000
	artificial, partially exposed	0.0002	0.000			0.500			0.000			0.000
	artificial, protected	0.0001	0.000	0.000	0.000	0.000			0.000		0.000	1.000
Plunging Rocky Shore	bedrock, exposed	0.0048	0.265	0.000		0.469	0.102	0.163	0.653			0.000
	bedrock, partially exposed	0.0115	0.214	0.068		0.359	0.068	0.120	0.598			0.009
	bedrock, semi-protected	0.0134	0.360	0.000		0.015		0.066	0.912		0.000	0.000
	bedrock, protected	0.0149				0.283						
	boulder, partially exposed	0.0018				0.500			0.333			
	boulder, semi-protected	0.0005							0.800			
	boulder, protected	0.0010				0.100						
	mixed coarse, partially exposed	0.0025										0.040
	mixed coarse, semi-protected	0.0015	0.200	0.000	0.267	0.000	0.000	0.067	0.933	0.000	0.000	0.000
	mixed coarse, protected	0.0029	0.100		0.000	0.133	0.000	0.467	0.433	0.000	0.000	
	gravel, partially exposed	0.0006	0.333	0.167	0.000	0.333	0.000	0.000	0.333	0.000		0.000
	gravel, semi -protected	0.0007	0.000		0.000	0.000		0.000	1.000			
	gravel, protected	0.0004	0.250	0.000	0.000	0.000		0.000	0.750			0.000
	sand, exposed	0.0001	0.000		0.000	1.000	0.000	0.000	0.000			
	sand, partially exposed	0.0003	0.333			0.333		0.333	0.667			
	sand, semi -protected	0.0003				0.333		0.000	1.000			
	sand, protected	0.0001	0.000	0.000		0.000	0.000	0.000	0.000			0.000
	mixed fine, partially exposed	0.0021	0.000		0.095			0.571	0.000			
	mixed fine, semi -protected	0.0001	0.000			0.000		0.000	0.000			

	mixed fine, protected	0.0028	0.034	0.138		0.345			0.276			
	mud, partially exposed	0.0001							0.000			
	mud, semi-protected	0.0001	0.000						0.000			
	mud, protected	0.0001				0.000						
	artificial, semi-protected	0.0001	1.000			0.000	0.000	0.000	1.000			
	artificial, protected	0.0003				0.333			0.000			
Rocky Platform/ramp	bedrock, exposed	0.0173										
	bedrock, partially exposed	0.0339		0.093	0.043	0.446	0.006	0.035				
	bedrock, semi-protected	0.0275										
	bedrock, protected	0.0359	0.104	0.022		0.156		0.321	0.696			
	boulder, partially exposed	0.0038				0.513						
	boulder, semi-protected	0.0007				0.286						
	boulder, protected	0.0020										
	cobble, partially exposed	0.0005	0.400						1.000			
	mixed coarse, partially exposed	0.0218				0.387		0.095				
	mixed coarse, semi-protected	0.0085	0.253	0.023	0.011	0.011	0.000	0.149	0.839	0.000	0.000	0.011
	mixed coarse, protected	0.0161	0.037	0.000	0.012	0.232	0.000	0.311	0.579	0.000	0.000	0.018
	gravel, exposed	0.0001	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
	gravel, partially exposed	0.0022	0.182	0.364	0.182	0.364	0.000	0.000	0.409		0.227	0.273
	gravel, semi -protected	0.0012	0.083	0.000	0.000	0.000		0.167	0.917		0.000	0.000
	gravel, protected	0.0007	0.143			0.000		0.000	0.286			
	sand, exposed	0.0001	0.000	0.000		1.000	0.000		1.000			
	sand, partially exposed	0.0050						0.098	0.235			
	sand, semi -protected	0.0021	0.429		0.095				0.810	0.000		
	sand, protected	0.0023	0.000	0.043	0.000	0.261	0.000	0.000	0.217	0.000	0.000	0.043
	mixed fine, partially exposed	0.0099	0.000	0.198	0.040	0.644		0.238	0.030			0.129
	mixed fine, semi -protected	0.0021	0.143		0.000	0.048		0.095	0.286			0.000
	mixed fine, protected	0.0136										
	mud, partially exposed	0.0001	0.000	0.000	0.000	0.000			0.000		0.000	0.000
	mud, semi-protected	0.0008	0.000					0.000	0.000			
	mud, protected	0.0003	0.000						1.000			
	organic, semi-protected	0.0001				0.000			0.000			
	reef, partially exposed	0.0002	0.000	0.500	0.000	0.500	0.500	0.000	0.500	0.000	0.000	0.500
	artificial, partially exposed	0.0012	0.000	0.167	0.083	0.417	0.000		0.083		0.000	
	artificial, semi-protected	0.0002		0.000		0.500			1.000			0.000
	artificial, protected	0.0003		0.000	0.000	0.000	0.000	0.333	0.333	0.000	0.000	0.000
	Sum	1.0000										

Following pages:

Table D8. Correlation matrix for normalized values used in calculation of composite index. Green cell means |correlation| ≥ 0.75; orange cell means |correlation| ≥ 0.5; yellow cell means |correlation| ≥ 0.25; gray cell means |correlation| ≥ 0.1; minus sign means negative correlation. Negative correlations not indicated if |correlation| < 0.1.

Table D8. Correlation matrix for normalized values used in calculation of composite index.

	Abalone	Clam intertidal	Clam subtidal	Dungeness crab	Oyster	Pandalid Shrimp	Urchin	Geoduck	Herr ing Holding	Herring Spawning	Sand Lance	Surf Smelt
Abalone												
Clam intertidal	-											
Clam subtidal												
Dungeness Crab	-											
Oyster	-											
Pandalid Shrimp												
Urchin		-		-	-							
Geoduck	-						-					
Herring holding area												
Herring Spawning area							-					
Sand Lance	-						-					
Surf Smelt	-						-					
Bull Trout												
Chinook Salmon												
Chum Salmon												
Cutthroat							-					
Coho Salmon												
Pink Salmon												
Sockeye												
Steelhead Trout												
Audubon IBA												
Shorebird Concentration												
Waterfowl Concentration							-					
Bald Eagle Communal Roost												
Bald Eagle Nest												
Black Oystercatcher Nest												
Great Blue Heron Colony												
At Risk birds 2005 to 2009 (PSAMP)												
Medan bird density, 2000 to 2009							-					
Seal and Sea Lion Haul Outs												
National Wetlands Inventory	-					-	-					
dune grass (grasses)							-					
sedge (salt marsh)												
high salt marsh												
low salt marsh							-					
Laminaria (kelp)												
chocolate brown kelp												
surfgrass												
eelgrass							-					
bull kelp												
macrocystis kelp												

Table D8 (continued). Correlation matrix for normalized values used in calculation of composite index.

	Bull Trout	Chinook Salmon	Chum Salmon	Cutthroat	Coho Salmon	Pink Salmon	Sockeye	Steelhead Trout
Abalone								
Clam intertidal								
Clam subtidal								
Dungeness Crab								
Oyster								
Pandalid Shrimp								
Urchin				-				
Geoduck								
Herring holding area								
Herring Spawning area								
Sand Lance								
Surf Smelt								
Bull Trout								
Chinook Salmon								
Chum Salmon								
Cutthroat								
Coho Salmon								
Pink Salmon								
Sockeye								
Steelhead Trout								
Audubon IBA								
Shorebird Concentration								
Waterfowl Concentration								
Bald Eagle Communal Roost								
Bald Eagle Nest								
Black Oystercatcher Nest								
Great Blue Heron Colony								
At Risk birds 2005 to 2009 (PSAMP)								
Medan bird density, 2000 to 2009								
Seal and Sea Lion Haul Outs								
National Wetlands Inventory								
dune grass (grasses)								
sedge (salt marsh)								
high salt marsh								
low salt marsh								
Laminaria (kelp)								
chocolate brown kelp								
surfgrass								
eelgrass								
bull kelp								
macrocystis kelp								

Table D8 (continued). Correlation matrix for normalized values used in calculation of composite index.

	Audubon IBA	Shorebird Conc.	Waterfowl Conc.	Eagle Communal Roosts	Bald Eagle Nest	Oystercatcher Nests	Great Blue Heron Colonies	At Risk birds 2005 to 2009	bird density 2000 to 2009	Seal & Sea Lion Haulouts
Abalone										
Clam intertidal										
Clam subtidal										
Dungeness Crab										
Oyster										
Pandalid Shrimp										
Urchin			-						-	
Geoduck										
Herring holding area										
Herring Spawning area										
Sand Lance										
Surf Smelt										
Bull Trout										
Chinook Salmon										
Chum Salmon										
Cutthroat										
Coho Salmon										
Pink Salmon										
Sockeye										
Steelhead Trout										
Audubon IBA										
Shorebird Concentration										
Waterfowl Concentration										
Bald Eagle Communal Roost										
Bald Eagle Nest										
Black Oystercatcher Nest										
Great Blue Heron Colony										
At Risk birds 2005 to 2009 (PSAMP)										
Medan bird density, 2000 to 2009										
Seal and Sea Lion Haul Outs										
National Wetlands Inventory										
dune grass (grasses)										
sedge (salt marsh)										
high salt marsh										
low salt marsh										
Laminaria (kelp)										
chocolate brown kelp										
surfgrass										
eelgrass										
bull kelp										
macrocystis kelp										

Table D8 (continued). Correlation matrix for normalized values used in calculation of composite index.

	National Wetlands	dune grass (grasses)	sedge (salt marsh)	high salt marsh	low salt marsh	Laminaria (kelp)	chocolate brown kelp	surfgrass	eelgrass	bull kelp	macrocystis kelp
Abalone	-										
Clam intertidal											
Clam subtidal											
Dungeness Crab											
Oyster											
Pandalid Shrimp	-										
Urchin	-	-			-				-		
Geoduck											
Herring holding area											
Herring Spawning area											
Sand Lance											
Surf Smelt											
Bull Trout											
Chinook Salmon											
Chum Salmon											
Cutthroat											
Coho Salmon											
Pink Salmon											
Sockeye											
Steelhead Trout											
Audubon IBA											
Shorebird Concentration											
Waterfowl Concentration											
Bald Eagle Communal Roost											
Bald Eagle Nest											
Black Oystercatcher Nest											
Great Blue Heron Colony											
At Risk birds 2005 to 2009 (PSAMP)											
Medan bird density, 2000 to 2009											
Seal and Sea Lion Haul Outs											
National Wetlands Inventory											
dune grass (grasses)											
sedge (salt marsh)											
high salt marsh											
low salt marsh											
Laminaria (kelp)											
chocolate brown kelp											
surfgrass											
eelgrass											
bull kelp	-										
macrocystis kelp											

Appendix E:
Scientific Names of Animal Species Mentioned in Report

Common Name	Scientific Name	Non-native
	Invertebrates	
butter clam	*Saxidomus giganteus*	
geoduck	*Panopea abrupta*	
northern abalone	*Haliotis kamtschatkana*	
acorn barnacle	*balanus glandula*	
Pacific oyster	*Crassostrea gigas*	X
Olympia oyster	*Ostrea conchaphila*	
red sea urchin	*Strongylocentrotus franciscanus*	
green sea urchin	*Strongylocentrotus droebachiensis*	
pink shrimp	*Pandalus jordani, P. borealis*	
coonstripe shrimp	*Pandalus danae*	
spot shrimp	*Pandalus platyceros*	
Dungeness crab	*Cancer magister*	
Taylor's checkerspot butterfly	*Euphydryas editha taylori*	
	Fish	
Pacific lamprey	*Entosphenus tridentate*	
Pacific sand lance	*Ammodytes hexapterus*	
surf smelt	*Hypomesus pretiosus*	
Pacific herring	*Clupea pallasi*	
chum salmon	*Oncorhynchus keta*	
pink salmon	*Oncorhynchus gorbuscha*	
Chinook salmon	*Oncorhynchus tshawytscha*	
sockeye salmon	*Oncorhynchus nerka*	
kokanee	*Oncorhynchus nerka*	
coho salmon	*Oncorhynchus kisutch*	
steelhead trout	*Oncorhynchus mykiss*	
rainbow trout	*Oncorhynchus mykiss*	
cutthroat trout	*Oncorhynchus clarki*	
coastal cutthroat trout	*Oncorhynchus clarki clarki*	
bull trout	*Salvelinus confluentus*	
	Amphibians	
western toad	*Anaxyrus boreas*	
	Reptiles	
none		
	Birds	
tundra swan	*Cygnus columbianus*	
trumpeter swan	*Cygnus buccinator*	
Canada goose	*Branta canadensis*	
snow goose	*Chen caerulescens*	
harlequin duck	*Histrionicus histrionicus*	
white-winged scoter	*Melanitta fusca*	
bald eagle	*Haliaeetus leucocephalus*	
peregrine falcon	*Falco peregrines*	
great blue heron	*Ardea herodias*	
sandhill crane	*Grus Canadensis*	

Common Name	Scientific Name	Non-native
	Birds	
black oystercatcher	*Haematopus bachmani*	
rock pigeon	*Columba livia*	X
band-tailed pigeon	*Columba fasciata*	
northern spotted owl	*Strix occidentalis caurina*	
pileated woodpecker	*Dryocopus pileatus*	
northern flicker	*Colaptes auratus*	
Pacific-slope fly catcher	*Empidonax difficilis*	
American crow	*Corvus brachyrhynchos*	
horned lark	*Eremophila alpestris*	
streaked horned lark	*Eremophila alpestris strigata*	
white-breasted nuthatch	*Sitta carolinensis*	
brown creeper	*Certhia Americana*	
winter wren	*Troglodytes troglodytes*	
American robin	*Turdus migratorius*	
Swainson's thrush	*Catharus ustulatus*	
western bluebird	*Sialia mexicana*	
cedar waxwing	*Bombycilla cedrorum*	
European starling	*Sturnus vulgaris*	X
house sparrow	*Passer domesticus*	X
house finch	*Carpodacus mexicanus*	
American goldfinch	*Carduelis tristis*	
white-crowned sparrow	*Zonotrichia leucophrys*	
song sparrow	*Melospize melodia*	
	Mammals	
Norwegian rat	*Rattus norvegicus*	X
eastern gray squirrel	*Sciurus carolinensis*	X
Douglas' squirrel	*Tamiasciurus douglasi*	
Olympic marmot	*Marmota Olympus*	
western pocket gopher	*Thomomys mazama*	
Mazama pocket gopher	*Thomomys mazama*	
mountain goat	*Oreamnos americanus*	
black-tailed deer	*Odocoileus hemionus columbianus*	
elk	*Cervus elaphus*	
raccoon	*Procyon lotor*	
red fox	*vulpes vulpes*	
black bear	*Ursus americanus*	
cougar	*Puma concolor*	
gray wolf	*Canis lupus*	
Keen's myotis	*Myotis evotis keenii*	
harbor seal	*Phoca vitulina*	
California sea lion	*Zalophus californianus*	

Appendix F: Data Dictionaries for GIS layers containing the Assessments' Results

Terrestrial Assessment			
FILE	**FIELD NAME**	**DESCRIPTION**	**UNIT or VALUE**
AU_Terrestrial_Indices_Aug2012	AU	Unique numeric identifier for original assessment unit (AU) polygon	integer
	New_AU	Unique numeric identifier for new assessment unit (AU) polygon. Not all original AU polygons could be mapped to new AU polygons. These records will have "NULL" values.	integer
	WRIA	unique numeric identifier for WRIA	integer
	Acres	area of AU polygon in acres	real
	Integ_Index	average value of landscape integrity index in AU	0 <= X<= 1
	norm_PHS	normalized score for PHS (priority habitat and species) data calculated for AU	0 <= X<= 100
	norm_oakgrass	normalized score for grassland-oak woodland data calculated for AU	0 <= X<= 100
	overall_INDEX	an index of relative conservation value which equals the maximum of Integ_Index*100, norm_PHS, norm_oakgrass	0 <= X<= 100
	vigin_freq	disctribution of Index divided into 20 quantiles (vigintiles) base on number of AUs. Each quantile contains 5% of AUs. May not have 20 levels because top two vigintiles have the same index value	1,2,3, . . . , 20
	decil_freq	disctribution of Index divided into 10 quantiles (deciles). Each quantile contains 10% of AUs. May not have 10 levels because top two deciles have the same index value	1,2,3, . . . , 10
	oct_freq	disctribution of Index divided into 8 quantiles (octiles). Each quantile contains 12.5% of AUs. May not have 8 levels because top two octiles have the same index value	1,2,3, . . . , 8
	vigin_area	disctribution of Index divided into 20 quantiles (vigintiles) based on AU area. Each quantile contains 5% of Puget Sound Basin area. May not have 20 levels because top two vigintiles have the same index value	1,2,3, . . . , 20
	MAX_type	INDEX euqals max of norm_PHS, norm_oakgrass, and Integ_Index*100. This shows which was maximum: integrity, PHS, or oakgress	character
	per_priv	percent of AU that is private land	0 <= X<= 100

Terrestrial Assessment			
FILE	**FIELD NAME**	**DESCRIPTION**	**UNIT or VALUE**
fragstats_impacts_zones_Aug2012	BLOCK	unique numeric identifier for open space block polygon	integer
	grid_code	unique numeric identifier for open space block; remant from raster version of this layer	integer
	ACRES	area of open space block	acres
	IMPACT_ext	index indicating degree of adverse impacts upon open space block from land uses outside the block; 1000 is maximum possible impact	0 <= X<= 1000
	IMPACT_int	index indicating degree of adverse impacts upon open space block from land uses inside the block; 1000 is maximum possible impact	0 <= X<= 1000
	HECTARES	area of open space block	hectares
	PROX	proximity index from FragStats	X > 0
	NORM_HECT	normalized area: NORM_HECT = area/benchmark; benchmark of 50,000 hectares	0 <= X<= 1
	NORM_CIRCLE	normalized circle (i.e., shape) index from FragStats	0 <= X<= 1
	NRM_IMP_int	normalized IMPACT_int; 0 is maximum possible impact	0 <= X<= 1
	NORM_PROX	normalized PROX	0 <= X<= 1
	NRM_IMP_ext	normalized IMPACT_ext; 0 is maximum possible impact	0 <= X<= 1
	internal	weighted geometric mean of NORM_HECT, NRM_IMP_int, and NORM_CIRCLE	0 <= X<= 1
	external	weighted geometric mean of NORM_PROC and NRM_IMP_int	0 <= X<= 1
	integ_index	weighted geometric mean of internal and external	0 <= X<= 1
	zone_index	weighted arthimetic mean of vegetation zone areas within block; there are 5 possible modified GAP vegetation zones	0 <= X<= 1
	overall_value	maximum of integ_index OR the weighted geometric mean of integ_index and zone_index	0 <= X<= 1

Terrestrial Assessment			
FILE	**FIELD NAME**	**DESCRIPTION**	**UNIT or VALUE**
AU_PHS_percentages_Sep2012	AU	Unique numeric identifier for original assessment unit (AU) polygon	integer
	New_AU	Unique numeric identifier for new assessment unit (AU) polygon. Not all original AU polygons could be mapped to new AU polygons. These records will have "NULL" values.	integer
	Audubon	Audubon important bird areas (IBAs): maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Bald_Eagle	bald eagle cummunal roosts: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Elk	maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Gopher	mazama pocket gopher: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Harlequin_Duck	Harlequin riverine habitat: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Peregrine_Falcon	peregrine falcon overwintering habitat: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Sandhill_Crane	maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Seabird_Conc	maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Shorebird_Conc	maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Horned_Lark	streaked horned lark habitat: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100

	Swan	swan overwintering habitat: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Taylor_Checkerspot	Taylor's checkspot butterfly habitat: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	Waterfowl_Conc	maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	oakgrass	oak woodlands and prairies from NHP: maximum of (percent of AU area covered by habitat) OR (percent of region's total habitat area contained in AU)	0 <= X<= 100
	max_species	name of species that has maximum percent value in the AU	character

Freshwater Assessment			
FILE	**FIELD NAME**	**DESCRIPTION**	**UNIT or VALUE**
	WRIA	Unique numeric identifier for water resources inventory area	integer
AU_freshwater_indices_Nov2013	OldAU	Unique numeric identifier for original assessment unit (AU) polygon	integer
	New_AU	Unique numeric identifier for new assessment unit (AU) polygon. Not all original AU polygons could be mapped to new AU polygons. These records will have "NULL" values.	integer
	AU_AC	area of AU polygon in acres	real
	Ecol_Integ	aquatic ecological integrity within AU	0 <= X<= 100
	sum_sumHU	sum within AU of reach-level summed habitat units (i.e., sum of salmonid species' habitats); values ranked and normalized by WRIA. See report for details (The Puget Sound Watershed Characterization Project Volume 2: A Coarse-scale Assessment of the Relative Value of Small Drainage Areas and Marine Shorelines for the Conservation of Fish and Wildlife Habitats in Puget Sound Basin).	0 <= X<= 1
	sum_maxHV	sum within AU of reach-level habitat values greater than 80th percentile habitat value in WRIA; values ranked and normalized by WRIA. See report for details.	0 <= X<= 1
	WHI_NoStN	waterhed habitats index (local salmonids index) within AU, does not include species ESA and SaSI status; values ranked according to number of AUs and normalized by WRIA	0 <= X<= 1
	WHI_NoStA	waterhed habitats index (local salmonids index) within AU, does not include species ESA and SaSI status; values ranked according to area of AUs and normalized by WRIA	0 <= X<= 1
	WHI_Stat	waterhed habitats index: max(sum_sumHU, sum_maxHV), includes species ESA and SaSI status; values ranked according to number of AUs and normalized by WRIA	0 <= X<= 1
	flood_wetN	proportion of AU that is extant wetlands and undeveloped floodplains (hydrogeomorphic feature index); values ranked according to number of AUs and normalized by WRIA	0 <= X<= 1
	flood_wetA	proportion of AU that is extant wetlands and undeveloped floodplains(hydrogeomorphic feature index); values ranked according to area of AUs and normalized by WRIA	0 <= X<= 1
	acum_NoStN	summation of all reach-level habitat indices (RHIs) downstream from AU. Values ranked according to number of AUs and normalized by WRIA.	0 <= X<= 1
	acum_NoStA	summation of all reach-level habitat indices (RHIs) downstream from AU. Values ranked according to area of AUs and normalized by WRIA.	0 <= X<= 1
	Q10_WHIns	distribution of WHI divided into 10 quantiles (deciles); status not included. Each quantile contains 10% of AUs.	0,1,2,3, . . . , 10
	Q20_W	distribution of WHI divided into 20 quantiles (vigintiles); status	0,1,2,3, . . .

	HIns	not included. Each quantile contains 5% of AUs.	, 20
	Q10_WHIws	distribution of WHI divided into 10 quantiles (deciles); status included. Each quantile contains 10% of AUs.	0,1,2,3, . . . , 10
	Q20_WHIws	distribution of WHI divided into 20 quantiles (vigintiles); status included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	Q10_fldwet	distribution of hydrogeomorphic feature index divided into 10 quantiles (deciles); status included. Each quantile contains 10% of AUs.	1,2,3, . . . , 10
	Q20_fldwet	distribution of hydrogeomorphic feature index divided into 20 quantiles (vigintiles); status included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	Q10accumNs	distribution of accumulative downstream habitats index divided into 10 quantiles (deciles); status not included. Each quantile contains 10% of AUs.	1,2,3, . . . , 10
	Q20accumNs	distribution of accumulative downstream habitats index divided into 20 quantiles (vigintiles); status not included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	Q10accumWs	distribution of accumulative downstream habitats index divided into 10 quantiles (deciles); status included. Each quantile contains 10% of AUs.	1,2,3, . . . , 10
	Q20accumWs	distribution of accumulative downstream habitats index divided into 10 quantiles (vigintiles); status included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	A3ns_avg	average of WHI, accum downstream habitats, and hydrogeomorphic features; status not included in WHI and accum downstream habitats	1,2,3, . . . , 10
	A3ns_max	maximum of WHI, accum downstream habitats, and hydrogeomorphic features; status not included in WHI and accum downstream habitats	1,2,3, . . . , 20
	A3ns_Q20av	distribution of A3ns_avg divided into 20 quantiles (vigintiles); status not included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	A3ns_Q20mx	distribution of A3ns_max divided into 20 quantiles (vigintiles); status not included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	A3ws_Q20av	distribution of A3ws_avg divided into 20 quantiles (vigintiles); status included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	A3ws_Q20mx	distribution of A3ns_max divided into 20 quantiles (vigintiles); status included. Each quantile contains 5% of AUs.	1,2,3, . . . , 20
	Shape_Length	perimeter of AU polygon in feet	real
	SHape_Area	area of AU polygon in square feet	real

Marine Shoreline Assessment				
"Results_Group" File Geodatabase: Feature Classes and Fields Each data set comprises both polygon and arc topology feature classes, noted by a trailing "_poly" or "_arc" in its name.				
Feature Class Pair Name	**FIELD NAME**	**DESCRIPTION**	**UNIT or VALUE**	**Source**
These fields are common to all feature classes within the "Results_Group" file geodatabase.	OBJECTID	Internal ID generated by and for the GIS software's tracking	integer	ESRI
	Shape	Type of feature class (line/arc or polygon)	charact er	ESRI
	ShoBas_I D	unique numeric identifier for shoreline segment	integer	WDFW
	SZUnit_I D	Unique ID from DNR Shorezone feature class, retained to enable joins with the original Shorezone table.	integer	DNR
	P3SPU1	PSNERP Shoreline Process Unit 1 ID number from version 3 data.	integer	PSNERP
	P3SPU2	PSNERP Shoreline Process Unit 2 ID number from version 3 data.	integer	PSNERP
	New_SA U	PSNERP Shoreline Accounting Unit ID number, enhanced beyond version 3 data. The original PSNERP data did not include deltas. Deltas have been integrated by WDFW. New SAUs within deltas were assigned and a few revisions to SAUs adjacent to deltas were modified by WDFW in cooperation with the US Army Corps of Engineers.	integer	PSNERP /WDFW
	Beach_ft	Total beach length within one complete ShoBas segment.	double	GIS@W DFW
	Shape_Le ngth	Perimeter of polygon or length of arc, depending on feature class topology	double	ESRI
	Shape_Ar ea	Area of polygon (polygon feature classes only)	double	ESRI
index_window_a vg_2miles	segments	number of shoreline segments in 2 mile window; if =0, then swegemtn longer than window width	integer	WDFW
	Avg_Inde x	average of SumAll index within 2 miles window centered on segment	0 <= X<= 1	WDFW
	Norm_In dex	Avg_Index normalized to maximum value within oceanographic sub-basin	0 <= X<= 1	WDFW
	Basin	code for PSNERP oceanographic sub-basin	charact er	WDFW
PSNERP_Recom mendations	Rcm1_Be ach	Recommendations for SPU1 beaches	charact er	PSNERP
	Rcm1_E mbay	Recommendations for SPU1 embayments	charact er	PSNERP
	Rcm1_Inl	Recommendations for SPU1 inlets	charact	PSNERP

	et		er	
	Rcm2_Beach	Recommendations for SPU2 beaches	character	PSNERP
	Rcm2_Embay	Recommendations for SPU2 embayments	character	PSNERP
	Rcm2_Inlet	Recommendations for SPU2 inlets	character	PSNERP
ShoBas_index_results_NOrank_Sum_April2012	SumAll	an index of relative habitat value which equals the sum of normalized values for all biological data included in the assessment	0 <= X<= 1	WDFW
	AvgTop5	an index of relative habitat value which equals the sum of the 5 largest normalized values in that shoreline segment	0 <= X<= 1	WDFW
	vig_Sumll	SumAll index divided into 20 quantiles (vigintiles). Each quantile contains 5% of shoreline length in that oceanographic subbasin	1,2,3, . . . , 20	WDFW
	ter_SumAll	SumAll index divided into 3 quantiles (terciles). Each quantile contains 33% of shoreline length in that oceanographic subbasin.	1, 2, 3	WDFW
	vig_AvgTop5	AvgTop5 index divided into 20 quantiles (vigintiles). Each quantile contains 5% of shoreline length in that oceanographic subbasin	1,2,3, . . . , 20	WDFW
	Large_1	largest normalized value in the shoreline segment	0 <= X<= 1	WDFW
	Large_2	2nd largest normalized value in the shoreline segment	0 <= X<= 1	WDFW
	Large_3	3rd largest normalized value in the shoreline segment	0 <= X<= 1	WDFW
	Large_4	4th largest normalized value in the shoreline segment	0 <= X<= 1	WDFW
	Large_5	5th largest normalized value in the shoreline segment	0 <= X<= 1	WDFW
	TaxaHab_1	species, species group, or habitat corresponding to largest normalized value in the shoreline segment	character	WDFW
	TaxaHab_2	species, species group, or habitat corresponding to 2nd largest normalized value in the shoreline segment	character	WDFW
	TaxaHab_3	species, species group, or habitat corresponding to 3rd largest normalized value in the shoreline segment	character	WDFW
	TaxaHab_4	species, species group, or habitat corresponding to 4th largest normalized value in the shoreline segment	character	WDFW
	TaxaHab_5	species, species group, or habitat corresponding to 5th largest normalized value in the shoreline segment	character	WDFW

For Index Results, two sets of feature classes exist: one pair for SPU1 beaches, another for SPU2 beaches, where "#" = 1 or 2				
SPU#_Index_results_April2012	SPU_Len	total length of PSNERP process unit (across all ShoBas segmets) in feet	real	WDFW
	Avg_SumAll	average of SumAll index within PSNERP process unit	0 <= X<= 1	WDFW
	Norm_Avg_SumAll	Avg_SumAll normalized to mazimum value within oceanographic sub-basin	0 <= X<= 1	WDFW
	vigintile	Norm_SumAll index divided into 20 quantiles (vigintiles). Each quantile contains 5% of shoreline length in that oceanographic subbasin	1,2,3, . . . , 20	WDFW
	tercile	Norm_SumAll index divided into 3 quantiles (terciles). Each quantile contains 33% of shoreline length in that oceanographic subbasin.	1, 2, 3	WDFW

Marine Shoreline Assessment				
FIELD NAME	**DESCRIPTION**	**UNIT or VALUE**	**SOURCE**	**ORIGINAL GIS SOURCE**
ShoBas_ID	ID unique to beach but common between	integer	calculated	PSAMP, PSNERP, DNR SZ
TideLevel	Polygon location relative to shore	text	WDFW	
SZUnit_ID	Unique ID from DNR Shorezone feature class	integer	WDNR	
P3SPU1	PSNERP Shoreline Process Unit 1 ID number		PSNERP version 3	PSNERP_V3p0_HARN.mdb
P3SPU2	PSNERP Shoreline Process Unit 2 ID number		PSNERP version 3	PSNERP_V3p0_HARN.mdb
P3Basin	PSNERP Basins, with a single basin chosen for those shorelines having two basins originally assigned	text	PSNERP version 3 / WDFW	PSNERP_V3p0_HARN.mdb
P3SubBasin	PSNERP Basins, original	text	PSNERP version 3 / WDFW	PSNERP_V3p0_HARN.mdb
P3C_Type	PSNERP Shoreforms, with deltas added	text	PSNERP version 3 / WDFW	PSNERP_V3p0_HARN.mdb
New_SAU	PSNERP Shoreline Accounting Unit ID number, with deltas added	integer	PSNERP version 3 / WDFW	PSNERP_V3p0_HARN.mdb
New_Cell	PSNERP drift cells, with deltas added	text	PSNERP version 3 / WDFW	PSNERP_V3p0_HARN.mdb
PSAMP Basin	PSEMP (formerly PSAMP) basins	text	WDWF	
Beach_ft	Length of progenitor shoreline	Linear feet	ShoBas_Intersected_arc	DNRSZ_NoAtts_arc intersect with ShoBas_Intersector_poly
Area_sqft	Area of ShoBas polygon	Square feet	ShoBas_Intersector_poly	ShoBas_Intersector_poly
Rte_...	Route information and statistics	various	WDFW	
BC_Class	British Columbia (BC) 'coastal class' or 'shoreline type'	integer	DNR Shorezone	DNR Shorezone
Deth_Lump	Dethier shoreline classification, Combined	integer	DNR Shorezone	DNR Shorezone
C_Type	Shoreform			PSNERP version 3
Abalone	Northern abalone	Square feet	Abalone!	GeoLib.DBO.abalone
Clam_Hard	Clams; intertidal hardshell	Square feet	Clam_Hard!	GeoLib.DBO.cla

				mhard
Clam_Subt	Clams; subtidal hardshell	Square feet	Clam_Subt!	GeoLib.DBO.clamsubt
Crab_Dun	Crab; dungeness	Square feet	Crab_Dun!	GeoLib.DBO.crab
Crab_RedRk	Crab; red rock	Square feet	Crab_RR!	GeoLib.DBO.crab
Oyster	Pacific Oyster	Square feet	Oyster	GeoLib.DBO.oyster
ShrimpPan	Pandalid Shrimp	Square feet	ShrimpPan!	GeoLib.DBO.shrmppan
Urchin	Red and Green Sea Urchins	Square feet	Urchin!	GeoLib.DBO.urchin
Geoduck	Commercial geoduck tracts	Square feet	Geoduck!	GeoLib.DBO.geoduck
HerrHold	Herring Holding Areas	Square feet	HerrHold!	GeoLib.DBO.HERRHOLD_SV
HerrSpwn	Herring Spawning Areas	Square feet	HerrSpwn!	GeoLib.DBO.HERRSPN_SV
SandLance	Sand ance spawning beaches	Linear feet	SandLance!	GeoLib.Forage_Fish_Surveys
SandLance_Pot	Potential-only beaches for Sand lance	Count of sites	SandLance_Pot!	GeoLib.Forage_Fish_Surveys
Smelt	Surf smelt spawning beaches	Linear feet	Smelt!	GeoLib.Forage_Fish_Surveys
Smelt_Pot	Potential-only beaches for Smelt	Count of sites	Smelt_Pot!	GeoLib.Forage_Fish_Surveys
Bull Trout	Bull Trout		FishDist!	GeoLib_FishDist
Chinook Salmon	Chinook Salmon		FishDist!	GeoLib_FishDist
Chum Salmon	Chum Salmon		FishDist!	GeoLib_FishDist
Coast Resident Cutthroat	Coast Resident Cutthroat	Count of Fish-Bearing Stream Mouths within SAU	FishDist!	GeoLib_FishDist
Coho Salmon	Coho Salmon		FishDist!	GeoLib_FishDist
Kokanee Salmon	Kokanee Salmon		FishDist!	GeoLib_FishDist
Pink Salmon	Pink Salmon		FishDist!	GeoLib_FishDist
Rainbow Trout	Rainbow Trout		FishDist!	GeoLib_FishDist
Sockeye	Sockeye		FishDist!	GeoLib_FishDist
Steelhead Trout	Steelhead Trout		FishDist!	GeoLib_FishDist

Audubon	Audubon's bird polygons	Square feet	Audubon!	L:\lu_planning\PSP\psp_terrestrial\finefilter\audubon_bird_areas.shp
Hqn_Duck	Harlequin Duck	Square feet	Hqn_Duck!	GeoLib.DBO.PHSREGION_SV
Shorebirds	Large regular concentrations	Square feet	ShoreBirds!	GeoLib.DBO.PHSREGION_SV
Waterfowl	Large regular concentrations	Square feet	Waterfowl!	GeoLib.DBO.PHSREGION_SV
BE_ComRoost	Bald Eagle 1320 ft Communal Roosts zones	Square feet	BE_ComRoost!	GeoLib_baldeagle_bf / 1320
BE_Nest	Bald Eagle 800 ft Nest zones	Square feet	BE_Nest!	GeoLib_baldeagle_bf / 800
HABA_Nests	Black Oystercatcher nests	Count	HABA_Nests!	GeoLib_WS_OCCURPOINT_SV
GBH_600ft	Great Blue Heron 600-ft nest zones	Square feet	GB_Heron!	GeoLib_WS_OCCURPOLYGON_SV
AtRisk2005_2009 *	Sum of individual at risk birds* over the years 2005 to 2009	Sum	PSAMP_Bird_Summary!	PSAMP_bird_summary.csv
MedDens2000_2009	Median density of birds over the years 2000 to 2009, inclusive	birds/ha	PSAMP_Bird_Summary!	PSAMP_bird_summary.csv
AvgDens2000_2009	Average density of birds over the years 2000 to 2009, inclusive	birds/ha	PSAMP_Bird_Summary!	PSAMP_bird_summary.csv
HaulOuts	Seal haul-out points	Count	HaulOuts!	GeoLib.DBO.HaulOuts_SSL
ESTUARINE INTERTIDAL	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
LACUSTRINE LIMNETIC	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
LACUSTRINE LITTORAL	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
MARINE INTERTIDAL	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
PALUSTRINE	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
RIVERINE TIDAL	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
RIVERINE UPPER PERENNIAL	National Wetlands Inventory	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
Natl_Wetlands_Total	National Wetlands total of above	Square feet	Natl_Wetlands!	GeoLib.DBO.NWIPOLY_SV
Kelp	Giant and Bull Kelp beds	Square feet	Kelp!	GeoLib.DBO.phskelp

GRA_UNIT	Seaweed presence - null/continuous/partial 0 = null, 1 - partial, 2 = continuous	Coverage of Shorezone (Rank): 0 = Absent 1 = 0-50% 2 = 50% +	DNR_Shorezone	DNR Shorezone
SED_UNIT			DNR_Shorezone	DNR Shorezone
TRI_UNIT			DNR_Shorezone	DNR Shorezone
SAL_UNIT			DNR_Shorezone	DNR Shorezone
OYS_UNIT			DNR_Shorezone	DNR Shorezone
SBR_UNIT			DNR_Shorezone	DNR Shorezone
SAR_UNIT			DNR_Shorezone	DNR Shorezone
CHB_UNIT			DNR_Shorezone	DNR Shorezone
SUR_UNIT			DNR_Shorezone	DNR Shorezone
ZOS_UNIT			DNR_Shorezone	DNR Shorezone
NER_UNIT			DNR_Shorezone	DNR Shorezone
MAC_UNIT			DNR_Shorezone	DNR Shorezone

* At risk birds: marbled murrelet, peregrine falcon, bald eagle, great blue heron, common loon, brown pelican, common murre, and Brandt's cormorant.