Improving Methods for Regional Marine Conservation Assessments: Examples from the Pacific Northwest

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NOAA Coastal Services Center
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Foreword

The mission of The Nature Conservancy is to preserve the plants, animals, and natural communities that represent the diversity of life on Earth by protecting the lands and waters they need to survive. Recognizing that a focus on the marine environment is critical to achieving our mission, TNC has made an organizational commitment to expand its marine conservation efforts nationally and internationally.

In the sea as on the land, the first step in The Nature Conservancy's conservation approach is to identify the most important areas for conservation of biodiversity through a participatory, data-driven ecoregional assessment process. And, as would be expected, there are numerous challenges to, as well as substantial benefits of adapting our eco-regional planning methods to the marine environment.

The work that has been launched through The Conservancy's partnership with NOAA's Coastal Services Center has allowed us to make substantial progress in tackling these challenges. Through our work in the Pacific Northwest Coast Ecoregion, we have developed a benthic habitat model for the offshore areas to predict where biodiversity values are likely to be high. We have also been able to develop spatially-explicit threat analyses and refine our methodologies for the better integration of conservation and management across land and sea. We look forward to a continued partnership with the Coastal Services Center to address the many challenges that are still outstanding.

As detailed in the report, the ecoregions in the Pacific Northwest continue to hold significant examples of temperate ecosystems and species including seagrasses, salt and brackish marshes, and native shellfish reefs, even though in some cases they are severely degraded. The benthic habitat on the continental shelf, while heavily impacted, also contains important opportunities for conservation. Based on the best information currently available, we have identified portfolios of priority areas for conservation and management throughout the Pacific Northwest.

The identification of these priority conservation areas makes no presumption about the best strategies for conservation in these areas. The Conservancy will work with our partners to better understand the present and future threats to marine diversity, as well as the biological, socioeconomic, and political circumstances at each site. No single strategy works everywhere, and at any site multiple strategies will be needed. We hope that our assessments in the Pacific Northwest will help shape a new vision for and commitment to the successful conservation and management of coastal and marine ecosystems in the region. We also hope it will reinforce the many outstanding conservation activities already under way in the region and provide an impetus for new ones. The Conservancy plans to use these assessments to guide our own coastal and marine work in the ecoregion – to forge new partnerships and to design new conservation strategies.

Lynne Zeitlin Hale
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Preface

Over the course of the past several years, The Nature Conservancy and partners have worked on assessments in ecoregions throughout the entire Pacific Northwest from Oregon to Alaska. Marine considerations have figured prominently in the ecoregional assessments for the Pacific Northwest Coast, Puget Sound, Georgia Basin, Central and North Coasts of British Columbia, Southeast Alaska, and Cook Inlet. From these efforts in the Pacific Northwest we have developed new approaches and tools for marine ecoregional assessments. The purpose of this report is to document many of these innovations in one place. All of the ecoregional assessments represented in this report are now available in their entirety elsewhere (http://www.conserveonline.org/workspaces/MECA).

We have divided this report into six sections. Each section is designed so that is both a cohesive part of this comprehensive report, and a stand alone product. In the first and second sections of this report we provide information on the main marine elements of marine regional planning and the Pacific Northwest Coast ecoregion, specifically an area that covers land and sea across the coasts of Oregon, Washington and parts of British Columbia. This represents the most recently completed marine ecoregional assessment and, in addition to the innovations represented generally in this effort, it provides a solid example of the marine assessment process and products.

In the third section we illustrate and compare methods for developing habitat models in offshore benthic habitats, where direct data on species and ecosystems is scarce. In these offshore benthic environments, we must often use geophysical data (e.g.,geomorphology, depth and sediment type) to describe the physical environments that are most likely correlated with differences in biotic assemblages (e.g., plant and animal communities). We provide examples from the Northwest Coast ecoregion on how to develop the models.

In the fourth section we describe new methods for better assessing threats in coastal and marine environments. There is a lack of quantitative spatial analyses addressing the problem of land and marine-base threats for integrated terrestrial and marine planning. We have mapped and combined specific threats in the Pacific Northwest Coast ecoregion into a single index where those threats occur and utilized a decision support tool for evaluating the effects of different indices on priority sites selected for conservation.

In section five we describe innovations in integrated planning across the land-sea interface. Fully integrated regional planning is currently rare but it can improve the ecological accuracy and economic efficiency of our efforts in conservation and management. The need for efficiency and integration of effort across environments is greater in the coastal zone than anywhere else, because demand for these environments is high and conservation and management in these environments is more expensive than elsewhere. Integrated planning is not however a panacea and there are pitfalls to be avoided, which we illustrate. We use examples from the Puget Sound and Cook Inlet ecoregions to illustrate these concepts.

Beyond the technical challenges, effective marine conservation requires that those involved in decision-making be engaged and "buy in" to the planning process. Through this work, we have been able to strengthen our consultation and partnership process with critical resource management agencies in the Pacific Northwest including NOAA's Coastal Services Center; and with them, we have been able to set initial conservation priorities. Perhaps of even greater importance is we now have a shared, easily accessible, spatial database that includes substantial information on both key marine ecosystems and threats that can be used to inform a range of decisions that will need to be over the coming years.

The advances that have been made possible through this partnership are an important step in developing both the tools and relationships that are essential for advancing the conservation and sustainable use of marine systems in the Pacific Northwest. This work will also inform the large number of other ecoregional assessments that The Conservancy is carrying out both in the US and internationally.

Acknowledgements

The Global Marine Initiative of The Nature Conservancy (TNC) would like to thank the National Oceanic & Atmospheric Administration's Coastal Services Center (NOAA CSC) for funding to build regional planning innovations and refine database design techniques into the marine ecoregional planning process in the Pacific Northwest. This partnership project was conducted through NOAA's Coastal Service Center, GIS Integration and Development Program. The goal of the partnership was to improve methods for identifying priority sites for marine conservation and management by integrating across environments in the Pacific Northwest.

This project was funded by an 18-month cooperative agreement to develop methods that efficiently identify priority sites for conservation and management action in coastal zones through spatially explicit regional planning that unambiguously accounts for the vital connections between land, rivers, and sea. The project has provided an integrated information resource on the distributions of key species, habitats and ecosystems across the Pacific Northwest. In addition to developing an integrated resource of spatial information on biological distributions in nearshore marine environments, this project has integrated information on the distributions of threats to coastal natural resources from terrestrial and freshwater environments. This partnership project has developed techniques, methods, and approaches to selecting high priority conservation areas both at sea and on land in an environmentally integrated manner.

This project was also made possible by the significant contributions from many people across The Nature Conservancy, our partner groups and agencies, and external reviewers.

1.0 Introduction – approach to regional planning

The Nature Conservancy uses a systematic regional planning approach for all of its assessments. Scientists, agencies, and private organizations are increasingly using systematic approaches to identify where and how to allocate conservation efforts, particularly at the regional level (e.g., Possingham et al. 1999; Day and Roff 2000; Leslie et al. 2002; Airame et al. 2003). The Nature Conservancy has been among those at the forefront in the development of new approaches for systematic regional planning (e.g., Beck and Odaya 2001; Groves et al. 2002; Groves 2003; Beck 2003, Ferdaña 2005).

Our approach to systematic planning is to:

1. Identify objectives, which in our assessments is to identify and represent a full range of the region's biodiversity for conservation;
2. Select targets (e.g., species and ecosystems) to represent this biodiversity and be the focus of conservation efforts;
3. Identify goals for the amount (abundance, area, distribution) of the targets required to meet objectives;
4. Identify suitability factors (e.g., human population density, shipping lanes) likely to affect either the cost of conservation, the viability of targets in any area, or the suitability of a specific area for conservation;
5. Develop a spatial database from all the reasonably available regional-scale data on the targets and suitability factors; and
6. Establish stratification and planning units in which the distribution of targets and suitability factors are tracked.
7. Select priority conservation areas to achieve the stated goals and objectives. Site-selection tools are commonly used to help process this information towards optimal solutions that meet objectives. In this assessment, the software program MARXAN, and it's precursors, (Ball and Possingham 2000) were used to arrive at multiple potential solutions. The results from MARXAN are peer-reviewed and modified in workshops with scientists, managers, and conservation practitioners to develop a final portfolio of conservation areas.

2.0 Pacific Northwest Coast marine ecoregional assessment

In the Pacific Northwest, The Nature Conservancy has teamed with the Washington Department of Fish and Wildlife and Nature Conservancy Canada to develop ecoregional conservation assessments. This partnership has provided needed synergy for all aspects of crafting the assessments and will be invaluable for implementing them.

The partnering organizations and agencies mutually agreed on a process to assess the biodiversity of the Pacific Northwest Coast ecoregion and to analyze data to develop the conservation assessment portrayed in the following report. Each organization and agency in the partnership has contributed expertise and by so doing has created a more robust assessment useful to a broader community of conservation interests in the ecoregion. Each partner has benefited from this effort by developing stronger institutional ties with the other partners and by receiving specific analyses and products that meet their own planning needs.

Worldwide, the ever-increasing demands on natural resources require society to make important decisions about resource use and conservation. Society faces the critical challenge of making strategic investments in conservation to protect the planet's natural heritage while minimizing conflicts with the legitimate and unavoidable use of natural resources. Towards this end, The Nature Conservancy, in cooperation with key partners, make informed decisions about where these investments should be made by developing scientifically-rigorous conservation assessments for every North American ecoregion. These comprehensive assessments evaluate the full spectrum of biodiversity in a given ecoregion, identifying areas of biological significance where conservation efforts have the greatest potential success.

The Pacific Northwest Coast ecoregional assessment is the product of a partnership initiated in 2001 to identify priority conservation areas in this ecoregion. The Nature Conservancy (TNC), the Nature Conservancy of Canada (NCC), and the Washington Department of Fish and Wildlife (WDFW) are the primary partners in this project. NatureServe, the Oregon Natural Heritage Information Center (ONHIC), the Washington Natural Heritage Program (WNHP), Nearshore Habitat Program of the Washington Department of Natural Resources (WDNR), and the British Columbia Conservation Data Centre (CDC) were major contributors of technical expertise and data. The project has also benefited from the participation of many other scientists and conservation experts as team members and expert reviewers.

The purpose of this ecoregional assessment is to identify priority areas for conserving the biodiversity of the Pacific Northwest Coast ecoregion. This assessment is a guide for planners and decision-makers, and has no regulatory authority. We have conducted this work in a transparent manner and are making it accessible to the widest range of users possible. It should be treated as a first approximation, and the gaps and limitations described herein must be taken into consideration by users. This work was prepared with the expectation that it will be updated and benefit from these updates, as the state of scientific knowledge improves, methods are further refined, and other conditions change.

This assessment uses an approach developed by TNC (Groves et al. 2000, Groves et al. 2002) and other scientists to establish conservation priorities within the natural boundaries of ecoregions. Similar assessments have been completed for over 45 of the 81 ecoregions in the United States, and for several others outside the country, with the objective of completing assessments countrywide, and throughout the Americas, by 2008. The Nature Conservancy is leading many of these assessments, while others are led by partner organizations or agencies using the same basic methodology.

The goal for the Pacific Northwest Coast ecoregional conservation assessment is to:

Identify the suite of conservation areas that promote the long-term survival of all native plant and animal species and natural communities in the ecoregion.

This report documents the assessment process, including the steps taken to design the spatially-explicit 'conservation portfolio' for this ecoregion. It presents an ecoregion-wide assessment that identifies and prioritizes places of biological and conservation importance.

This section describes the assessment of ecological systems and species for the marine nearshore component of the Pacific Northwest Coast ecoregion. The purpose of the assessment was to develop a portfolio of priority conservation areas that, if conserved, will protect a representative subset of the nearshore marine biodiversity. The Northwest Coast marine assessment covers all shoreline and estuarine areas. We define the shoreline and estuarine environments as the "nearshore zone," the area extending from the supratidal zone above the ordinary or mean high water line (i.e., the top of a bluff or the extent of a high salt marsh or dune grass community) to roughly the 10 meter depth below mean lowest low water. The assessment also addresses a few shoal areas and most of the offshore island for which data were available.

2.1 Geographic Setting

From a conservation assessment perspective, ecoregions are defined as "…relatively large areas of land and water that contain geographically distinct assemblages of natural communities". The Pacific Northwest Coast ecoregion is a narrow, elongated ecoregion lying to the west of the Coast Range mountains and stretching from the southern border of Oregon to the northern tip of Vancouver Island (Figure 2.9.1). The ecoregion includes nearly all of the Olympic Peninsula and most of Vancouver Island, British Columbia encompassing some 8,170,260 ha (30,900 square miles) of temperate rainforests, beaches and rocky intertidal zones, bays and estuaries, and coastal rivers. Although the ecoregion's elevation averages only 445 meters, the effect of the adjacent mountains, ocean intrusions, and glaciation in the northern half of the ecoregion has caused dramatic localized differences in climate, soils, and geology. The marine and estuarine environments of the outer coast add even greater diversity of communities and species. The ecoregion contains over 16,000 kilometers of streams and rivers, and includes the lower reaches of several major rivers whose headwaters lie in adjacent ecoregions.

An integral part of the Pacific Northwest Coast ecoregional assessment is the marine environment. We have added a marine region boundary to the terrestrial ecoregion that generally follows those identified by the NOAA NERRS program. These are biogeographically-based, determined primarily by the distribution of nearshore species and ecosystems (http://nerrs.noaa.gov/Background_Bioregions.html). TNC made some modifications to the NERRS system largely based on expert advice. In particular, we modified boundaries to better line up with terrestrial ecoregions for a more integrated land-sea analysis in the coastal zone. The boundaries that have been adjusted to line up with terrestrial ecoregions are those between (i) the Northwest Coast and the Central & Northern California regions and (ii) between the Northwest Coast and Puget Sound in the Strait of Juan de Fuca. The Pacific Northwest Coast marine region includes all shoreline, estuarine, and offshore areas down to 2,500 meters deep.

The outer coasts of Oregon, Washington and the West Coast of Vancouver Island in British Columbia offer a wide range of intertidal and subtidal marine diversity. From exposed rocky shores of the Pacific Ocean to protected estuarine systems, the Northwest Coast ecoregion encompasses over 9,000 kilometers of shoreline. In general, the region is characterized by large amounts of rain in places along the coast which contribute freshwater run-off and land-derived nutrients to the marine environment.

The coastal waters of the ecoregion were delineated into nine marine sub-regions based on British Columbia's definition and delineation of marine "ecosections" (Harper 1993). Ecosections are characterized as unique physiographic, oceanographic, and biological assemblages that are related to water depth and habitat (pelagic versus benthic). The nine coastal ecosections are: Queen Charlotte Sound and Strait along the outer waters of north Vancouver Island; Johnstone Strait in the inland seas of Vancouver Island; Vancouver Island Continental Shelf along the Island's West Coast; the Strait of Juan de Fuca along the shores of both British Columbia and Washington; two sections north and south of Pt. Grenville in Washington; and two sections north and south of Cape Arago in Oregon (Figure 2.9.2). We used the freshwater input from the Columbia River as an additional parameter in delineating these ecosections.

2.2 Technical Teams

In conducting our nearshore marine analysis and evaluating the site selection process we have relied on three marine technical teams assembled in Oregon, Washington, and British Columbia. These teams assisted in the design of the nearshore methodology, providing scientific and technical advice, and participating in the expert review process. These teams represent a variety of state and federal agencies, universities, nonprofit organizations, and consulting firms.

Agencies and organizations that are represented in Oregon include:

Michele Dailey	Ecotrust
Cristen Don	ODFW Marine Resources Program
Tanya Haddad	DLCD Oregon Ocean-Coastal Management Program

Gayle Hansen	OSU Hatfield Marine Science Center
Steven S. Rumrill	ODSL South Slough National Estuarine Research Reserve
Maggie Sommer	ODFW Marine Resources Program

In Washington:

Helen Berry	WDNR Aquatic Resources Division
Philip Bloch	WDNR Aquatic Resources Division
Mary Lou Mills	WDFW Marine Resources Division

In British Columbia:

John R. Harper	Archipelago Marine Research Ltd.
Carol Ogborne	MSRM, British Columbia
Rob Paynter	MSRM, British Columbia
Mark Zacharias	MSRM, British Columbia

2.3 Selecting and Representing Nearshore Marine Targets

The nearshore marine technical teams and other experts identified 173 nearshore conservation targets comprising 84 coarse filter targets (58 shoreline types, 26 supratidal/intertidal/shallow subtidal ecosystems) and 89 fine filter targets (25 marine fish, 39 seabirds and shorebirds, 12 marine mammals, and 13 marine invertebrates). These targets were selected to represent nearshore marine biodiversity within the ecoregion, highlight threatened or declining species and communities (i.e., seabird colonies), or indicate the health of the larger ecosystem. Technical teams identified 18 additional area-based estuarine targets based on substrate type.

To recognize the unique ecological characteristics of outer coast, estuaries, and embayment environments, we stratified targets by coastal ecosections and further divided the shoreline ecosystem and intertidal targets into typological units within and outside of estuaries (Figure 2.9.3.).

2.3.1 Coarse Filter Targets

We have adopted the term "ecosystem" to describe plant communities in the nearshore environment after Beck et al. 2003. Nearshore ecosystems such as seagrass meadows, marshes and mangrove forests supply many vital ecological services in coastal waters, including shoreline protection, commercial and sport fisheries, and nutrient cycling. These ecosystems are considered nurseries for juvenile fish and shellfish. Where "ecosystem" is used to identify characteristic assemblages of plants and animals and the physical environment they inhabit, the term "habitat" refers to the area used by a species, with modifiers added to identify the particular habitats used by an animal. For example, blue crabs, *Callinectes sapidus*, utilize portions of seagrass and marsh ecosystems and we would refer to this as blue crab habitat. It should be noted that other scientific literature use the term "habitat type" to describe plant assemblages in the coastal zone (e.g., Morris 2001). In general we have modified the terminology to be consistent, though noting the differences from the scientific literature. For spatial representation of coarse filter targets, we have focused on spatial data development related to: (1) coastal ecosystems and habitats, (2) shoreline types, and (3) estuaries.

Coastal Ecosystems and Habitats

There were 26 individual supratidal, intertidal, and shallow subtidal ecosystems and habitats considered as conservation targets. Of these, we had spatial data for 11 of them. Six vegetated, coastal zone ecosystems were identified between the supratidal and the shallow subtidal: dune grasses (*Leymus mollis* and others*)*, saltmarshes (*Salicornia, triglochin, deschampsia,* and sedges), eelgrass (*Zostera*), surfgrass (*Phyllospadix*), algal beds (*Fucus* and mixed red algae) and kelps (*Macrocystis, nereocystis*). These ecosystems were analyzed in the assessment in both linear (shoreline) and area-based (estuaries) spatial formats (Figure 2.9.4). All of these ecosystems are either recognized to be ecologically important, known to be highly productive, or sensitive to human impacts. Although these categories alone do not represent the entire range of supratidal to shallow subtidal ecosystems or the most diverse ecosystems, they are believed to be good rough surrogates at the ecoregional scale. One additional ecosystem, rocky intertidal (termed "habitat type 3" in Morris 2001), was considered as a separate conservation target in this analysis. This habitat type or ecosystem can be identified in the lower intertidal by assembling indicator intertidal species on semi-exposed rocky shores (immobile substrates) including chocolate brown algae (*Hedophyllum, Egregia, L. setchellii, Eisenia*), California mussels (*Mytilus californianus),* surfgrasses (*Phyllospadix*), kelps (*Nereocystis*), and rich red algae beds (*Odonthalia* and others). This assemblage of plants and animals may have more likelihood of spatial diversity/heterogeneity and include specific habitats such as tidepools. We stratified these seven types by coastal marine ecosection, yielding 44 types (e.g., algal beds in the Strait of Juan de Fuca). We also used two typologies (outer coast and embayments) to separate the biological communities in and outside of estuaries. This brought the total number of unique types to 75 (e.g., algal beds in the outer coast of Strait of Juan de Fuca).

We utilized additional data sets illustrating areas of canopy kelps (*Macrocystis, nereocystis*) throughout the ecoregion, and eelgrass beds (*Zostera*) in the Strait of Juan de Fuca and Vancouver Island shelf ecosections. In Washington, existing floating kelp (*Macrocystis integrifolia*) planimeter data was collected along exposed coastline for the years 1989 to 2000. We created a kelp persistence index, adding all years together (1 thru 11) and classifying them into three distinct categories (1 to 3 years of kelp in class 1; 4 to 7 years in class 2; 8 to 11 years in class 3). These classes approximate the "observed once/outlier class," "regularly observed, but not always class," and the "always there/core sites class" and were analyzed as three spatial entities in an attempt to select the more persistent kelp beds. For Oregon and British Columbia areas of kelp beds were indicated as present when the surveys were done. In Oregon bull kelp (*Nerocystis leutkeana*) beds were photographed and mapped for the entire coast. Aerial photography occurred in summer of 1990. Kelp beds delineated off Cape Arago include giant kelp (*Macrocystis integrifolia*). *Macrocystis* was not found elsewhere on the Oregon coast. In British Columbia kelp and eelgrass beds were delineated off of Canadian Hydrographic Service charts. These data sets were left distinct from the kelp bed data in Washington. However, all types of spatial variation were considered as two conservation targets ("kelps" and "eelgrass"). Although we stratified these data by ecosection, we did not further divide them into outer coast and estuarine typologies because the data was considered to be mostly in the subtidal zone (deeper than zero Mean Lower Low Water or MLLW).

Shoreline Types

Shoreline types were derived from various classification systems. In general, we adopted the summary classification developed in British Columbia, the ShoreZone mapping system, and translated the various classifications to a single set of types. The Province of British Columbia developed its physical and biological ShoreZone mapping system based on shore types after Howes et al. (1994) and Searing and Frith (1995). Shore types are biophysical types that describe the substrate, exposure, and vegetation across the tidal elevation, as well as the anthropogenic features. There are 34 coastal classes and 17 representative types within the classification system. See Berry et al. (2001) for the rationale and definitions of the 34 coastal classes. We also considered the Dethier classification system (Dethier 1990) of intertidal communities and in Oregon NOAA's Environmental Sensitivity Index (ESI) classification based on combinations of substrate types in different sections of the intertidal zone (NOAA 1996). ESI combines substrate/morphology and wave energy, ranking the 23 coastal types according to oil spill sensitivity. There is not explicit mapping of biota within ESI.

We combined a derived version of British Columbia's representative shore types and ESI combinations into 15 shoreline ecosystems based on landform and slope (Figure 2.9.5). We then added an observed exposure, or fetch, type that was either derived directly from the data (ShoreZone) or calculated with a wave energy algorithm. For the Oregon coast we calculated fetch using a model developed by LTL Limited (Victoria, British Columbia, Canada). We did not use the wave energy attributes from ESI because not all shorelines were classified and many individual shoreline units contained multiple classes. These multiple wave energy classes (e.g., wave-cut platforms and exposed pier structures/ sheltered tidal flats) attempted to depict the landward to seaward shoreline characterization, but yielded too many combinations (41 unique classes as opposed to 17 representative types) and were therefore difficult to summarize. In addition, some classes did not make logical sense (e.g., wave-cut platforms and exposed pier structures/sheltered rocky shores and coastal structures) where the exposure type conflicted between landward and seaward types. We therefore stripped the exposure classes out of ESI and combined the substrate types with the representative shore types based primarily on landform. Next we added the observed and calculated exposure classes onto the seamless ecoregion-wide landform classes. Both the observed,maximum and effective fetch calculations were classified into four categories using Morris (2001). These included shorelines that were very exposed (VE), exposed (E), protected (P), and very protected (VP). Combining shore and exposure types yielded 58 shoreline ecosystem targets (Figure 2.9.6).

Our assumption was that by representing geomorphic and wave energy characteristics we would have a good idea of the biotic assemblages inhabiting them. In order to select these shoreline targets across the ecoregion we intersected them with the nine coastal ecosections. That calculation yielded 210 stratified targets (e.g., exposed sand flat in the Strait of Juan de Fuca). A further division was made in order to separate shorelines within and outside of estuaries. These typological units (outer coast shorelines, estuary shorelines) increased the number of unique shorelines to 304 (e.g., exposed sand flat in the outer coast of Strait of Juan de Fuca). Although we identified man-made and undefined shore types these types were not considered conservation targets.

Estuaries

An estuary (or embayment) is a zone of transition between the marine-dominated systems of the ocean and the upland river systems, a zone where the two mix, yields one of the most biologically productive areas on Earth (DLCD 1987). Delineation and characterization of estuaries, however, varies among researchers, agencies and geographies. Even among regional estuary mapping projects definitions and objectives of the mapping vary widely. We used four primary estuary mapping systems for this assessment, including: the British Columbia estuary mapping project from the Pacific Estuary Conservation Program (PECP); the ShoreZone mapping system in British Columbia (MSRM) and Washington (WDNR), the National Wetlands Inventory (NWI); U.S. Fish & Wildlife Service in both Washington and Oregon; and the Estuary Plan Book from Department of Land Conservation and Development (DLCD 1987) in Oregon.

The British Columbia estuary mapping project from PECP estimates the boundaries of an estuary using chart datums, water marks, and surface salinity intrusion referenced to specific spatial data. The intertidal zone features for each estuary system or complex were captured as polygons within the area found below the provincial Terrain Resource Inventory Mapping (TRIM) 1:20,000 coastline or island shoreline (< Mean higher high water mark) and above the zero chart datum contour line (> Lowest normal tide) depicted on Canadian Hydrographic Service (CHS) charts (Ryder et al. 2003). Although the PECP estuaries were delineated as polygons they did not contain any information on substrate characterization within them.

The ShoreZone mapping system conducted surveys at low-tide collecting aerial imagery during minus tides (below Mean lower low water) in June of the year (see Berry et al. 2001). Spatial data was based on the Washington Department of Natural Resources digital shoreline (water level line). For British Columbia, shore units were identified on the video and were transferred to 1:40,000 CHS charts for the west coast of Vancouver Island. In addition to delineating the features as polygons, ShoreZone attributed them with their dominant substrate type (e.g., organics/fines, sand flat, mud flat).

The National Wetlands Inventory (NWI) Database is an inventory system developed in 1974 by the U.S. Fish and Wildlife Service. Mapped at a scale of 1:24,000 or 1:62,000, NWI identifies wetlands and deep water habitats as either polygons or linear features. Attached to the mapped wetlands are descriptive codes based on the Cowardin classification system (Cowardin et al., 1979). NWI data is collected through stereoscopic analysis of high altitude color infrared aerial photographs. For Washington and Oregon the digital photography was done in the 1980s.

The Estuary Plan Book developed by the Department of Land Conservation and Development (DLCD 1987) in Oregon also delineates the extent of estuaries and substrate/vegetation polygons within them. Original base maps were prepared by the Division of State Lands in 1972 and 1973 using aerial photographs from the U.S. Geological Survey (USGS EROS Data Center, NASA). These base maps were used in 1978 and 1979 by the Oregon Department of Fish & Wildlife (ODFW) in its mapping of estuarine habitats as part of DLCD's estuary inventory project. The origins of both the delineation and characterization of these estuaries are from Cowardin et al. (1979) and modified by ODFW.

Given these variations for depicting estuaries we did not attempt to adopt a single definition of the extent of an estuary. Likewise we did not construct a single summary classification for substrate or vegetation types, but preserved them in our conservation target list. With this in mind, we combined the different data sets on extent and characterization into a single conservation assessment process. We collected spatial information on 187 estuaries in the ecoregion. There are 33 estuaries mapped on the Oregon coast (DLCD/NWI - approximately 89,281 hectares including all of Columbia River estuary), 16 mapped on the Washington coast (WDNR/NWI - approx. 67,016 hectares), and 138 mapped along the west coast Vancouver Island (CWS/MSRM/Ministry of Forests - approx. 8,345 hectares). Benthic substrate and vegetation types within these delineated estuaries were identified for 101,856 hectares out of a total 164,642 hectares (62%). Stated another way, we have benthic data contained within 89 out of 187 mapped estuaries (48%). Most of these data are contained within Oregon and Washington estuaries; smaller estuaries in British Columbia often lacked identified benthic types. The result was 18 substrate types (Figure 2.9.7). Some of these types were also represented in the shoreline type and coastal ecosystem targets (e.g., mud flat), but others were unique (e.g.., boulder). We decided to keep these additional area-based estuarine substrate targets separate from the shoreline and ecosystem targets, which are linear-based features (Figure 2.9.8).

2.3.2 Fine Filter Targets

Nearshore Marine Species

The marine technical teams selected species as fine filter targets generally following the criteria in Groves et al. (2002) and Beck et al. (2003). Workshops with regional experts resulted in a long list of species for consideration. When compiling species location data, we tried to compile data for the entire marine region. Coastal, nearshore, and offshore species were therefore considered. After evaluation of available data we decided to focus our efforts on coastal/nearshore species and treat offshore species in a later assessment.

The final list of nearshore marine conservation targets consisted of 89 species made up of 25 marine fish, 12 marine mammals, 39 seabirds/shorebirds, and 13 invertebrates. Of these targets we used spatial data representing 18 (two marine fish, one marine mammal, 13 seabirds/shorebirds, and two invertebrates), or 20%, of them in the analysis (Figure 2.9.9).

Forage Fish Spawning Beaches
Information was collected on two species of forage fish: Pacific Herring (*Clupea pallasi*) and Surf Smelt (*Hypomesus pretiosus*). Pacific Herring had comprehensive coverage in Washington (WDFW) and British Columbia (MSRM); Surf Smelt information was only available for Washington (WDFW). All spawning data was represented as presence of eggs on specific beach locations, although in both regions an absence of spawned eggs may mean a lack of survey effort rather than a true absence.

The Washington forage fish data represented historic and current spawning beaches over the last 10 years. This information is continually being updated and is not meant as a long-term indicator of presence or absence. The methods of data collection have steadily improved; therefore, updates are meant to augment older spawning locations. The data was collected on U.S. Geological Survey (USGS) maps at 1:24,000 then digitized into polygons for Pacific Herring and as linear features for Surf Smelt. We transformed the polygonal Herring data to linear features coinciding spatially with the ShoreZone mapping system in order to match a similar data set in British Columbia. We included historic site spawning locations for the analysis because of their known importance in the recent past. The Surf Smelt data was kept as a separate linear data set; the spatial extent of these data covered the Strait of Juan de Fuca on the Washington side, and the North and South Pt. Grenville ecosections.

The British Columbia Pacific Herring data was assembled as linear features using the same spatial shoreline as ShoreZone. Original attribute data indicated the Relative Importance (RI) of the feature per location. The RI values are only comparable within project regions (i.e., West Coast Vancouver Island) and not to other coastal zones in British Columbia. We selected RI values of 1and 2 to identify places of relatively low occurrence of Herring, and between 3 and 5 to identify relatively high occurrences. Presence of Pacific Herring were attributed to ShoreZone beach segments in a similar manner to those in Washington, allowing for a seamless identification of spawning beaches throughout the ecoregion.

Marine Mammal Haulout Sites
Steller sea lion (*Eumetopias jubatus*) haulout sites were the only marine mammal data included in this analysis. In Washington we utilized the atlas of seal and sea lion haul out sites (Jefferies et al. 2000) and a database (WDFW) showing current locations. For Oregon we received tabular location information (Robin Brown, personal communication 2003) that we made spatial as point features. In British Columbia we used point locations from University of British Columbia surveys; they distinguished haulout sites from rookery sites in the database, and we treated these as two spatial entities for one conservation target in the analysis.

Seabird Colonies and Shorebird Nesting Sites
We relied exclusively on seabird colony data in representing specific seabird species in the ecoregion. The Washington seabird colony database contains locations surveyed for breeding seabirds as documented in 'Catalog of Washington Seabird Colonies' by Speich and Wahl (1989). There were 18 species of seabirds listed as attributes in the colony

data, of which we identified 12 species as targets. These included Brandt's Cormorant (*Phalacrocorax penicillatus*), Cassin's Auklet (*Ptychoramphus aleuticus*), Caspian Tern (*Sterna caspia*), Common Murre (*Uria aalge*), Double-crested Cormorant (*Phalacrocorax auritus*), Fork-tailed Storm Petrel (*Oceanodroma furcata*), Leach's Storm Petrel (*Oceanodroma leucorhoa*), Pelagic Cormorant (*Phalacrocorax pelagicus*), Pigeon Guillemot (*Cepphus columba*), Rhinoceros Auklet (*Cerorhinca monocerata*), and Tufted Puffin (*Fratercula cirrhata*). An additional target, Black Oystercatcher (*Haematopus bachmani*), is a shorebird but is most often listed under seabird colony data sources. These species were also catalogued in Oregon from a USFWS database (surveys from 1979 to 2001). This seabird colony catalog contains 16 species; the same 12 target species found in Washington were matched in this database.

The British Columbia seabird colony inventory (Canadian Wildlife Service 2001) includes the locations of all known seabird colonies along the coast of British Columbia, and provides a compilation of the most recent (up to 1989) population estimates of seabirds breeding at those colonies. Fifteen species of seabirds, (including two storm petrels, three cormorants, one gull and nine alcids) and one shorebird (Black Oystercatcher *Haematopus bachmani)* breed along the coast of British Columbia. Over 5.6 million colonial birds are currently estimated to nest at 503 sites. Five species (Cassin's Auklets *Ptychoramphus aleuticus,* Fork-tailed Storm-petrels *Oceanodroma furcata,* Leach's Storm-petrels *Oceanodroma leucorhoa,* Rhinoceros Auklets *Cerorhinca monocerata, and* Ancient Murrelets *Synthliboramphus antiquus)* comprise the vast majority of that population, although Glaucous-winged Gulls (*Larus glaucescens)* and Pigeon Guillemots (*Cepphus columba)* nest at the most sites. All 12 seabird colony targets were represented in this database for the west coast and northern region of Vancouver Island.

One other seabird/shorebird target where we were able to gather spatial data was the Western Snowy Plover (*Charadrius alexandrinus nivosus*). These data represented significant point locations in Washington's (WDFW) priority habitats and species database, and polygonal data illustrating nesting sites and significant site locations during the breeding season (ODFW, ORNHIC) in Oregon. The point and polygon data sets remained as two distinct spatial entities for one conservation target. In addition, nesting and significant sites in the Oregon data were treated separately in the analysis, with the same target goal assigned to each distinct polygon feature type.

Intertidal Marine Invertebrates

Of the 13 invertebrates species recognized as conservation targets, we have assembled spatial data for only the mussels and barnacles (*Mytilus californianus - Semibalanus carious* with scattered *Pollicipes*). ShoreZone in both Washington (WDNR) and the west coast of Vancouver Island (MSRM) lumped mussel and barnacle observations as a single mid-intertidal species attribute. We treated these observations as two distinct targets.

There was much debate about what to consider an invertebrate conservation target based on the target selection criteria. There are many data gaps in our knowledge of invertebrate abundance, those that may be vulnerable regionally, those thought to be in decline, and those considered ecosystem engineers/keystone species. Although introduced shellfish, for example, can be ecosystem engineers and beneficial to the environment (i.e., filter feeding can cleanse the water column of toxins), they are never considered as conservation targets because they are non-native. As is often the case we simply did not have the information necessary to evaluate the status and condition of invertebrate communities, leaving us with a non-comprehensive list of invertebrate targets representing the region's diversity.

2.4 Data Gaps and Limitations

2.4.1 Coarse Filter Targets

The nearshore is subject to forces both oceanic and terrestrial, producing ecosystems that are dynamic and "open" in nature. This openness of marine populations, communities, and ecosystems probably has marked influences on their spatial, genetic, and trophic structures and dynamics in ways experienced by only some terrestrial species (Carr 2003). The nearshore is therefore not easily defined and mapped, making conservation planning more difficult than on land. Given that all data in a Geographic Information System (GIS) is represented at a specific time or limited time frame, and at a specific scale or resolution, there were inherent limitations in surveying the shoreline environment.

Although the ShoreZone mapping system is comprehensive in its representation of shoreline characteristics, we accepted some limitations when adopting this data set to develop our coarse filter targets. Tide, weather, visibility into the nearshore water column, and season all play important factors when conducting a shoreline inventory. Further, given the amount of shoreline in the ecoregion, these ShoreZone inventories had to be done over a period of years, and therefore survey methods were refined in later projects. We tried to account for this in the selection process, giving more weight to regions that had been surveyed more recently and thus contained better quality data. ShoreZone also does not distinguish between differences in the integrity of occurrences of the same ecosystem type. To some extent we compensated for this limitation by using data on shoreline modifications in the suitability index, so that the site selection model favored less altered sites. Updates to the shoreline inventories, therefore, need to occur at more frequent intervals, especially the biological component where species assemblages can dramatically change from year to year. Finally, there is inadequate data to represent the marine counterpart to terrestrial plant communities(i.e. associations of marine algae and sessile invertebrates). Likewise few algal species are adequately mapped across regions.

Mapping and characterization of estuaries varied throughout the ecoregion. This made it difficult to combine them into a single database or build a single, spatially defined set of estuarine conservation targets. Benthic substrate type definitions varied between Oregon, Washington and British Columbia estuaries, and often there was no characterization of them. This was also true of delineations of biological communities. Finally, the photographic imagery used to both delineate the boundaries of estuaries and identify substrate and biological communities within them was quite old. For instance, the Estuary Plan Book in Oregon is still considered the official estuary mapping product even though the base maps used are over 30 years old. Likewise the ShoreZone and National Wetlands Inventory mapping of estuaries varies, with most regions mapped 10 to 20 years ago. Given the fluctuation of conditions in estuaries and their degradation rates from dredging and development, we need more up-to-date estuary mapping products focused explicitly on biological assemblages.

The largest data gap in the nearshore is between five to 10 meters below Mean Lower Low Water (MLLW) and around 40 to 50 meters of water. This area, although surveyed either using multibeam or side-scan technologies at specific sites, has not been done regionally. This area is more labor intensive to survey, where more track lines need to be set to cover the same area as in deeper water. In addition, fisheries and fishery-independent surveys usually start at 50 meters or deeper. This gap is evident in regional vessel surveys conducted by NOAA, who have focused their attention on collecting multibeam information and conducting trawl surveys outside of bays, estuaries and the relative shallows of the coastal zone. There is clearly a need to comprehensively survey nearshore waters for benthic and biological factors, and utilize technologies such as LIDAR to construct more detailed nearshore bathymetry across larger areas.

2.4.2 Fine Filter Targets

Nearshore marine species data are either very coarse in scale (i.e., depicting a species' general distribution) or collected at very fine resolution (i.e., detailed survey transects a specific intertidal sites). Data sets were screened for inclusion in the regional analysis through an examination of data confidence and comprehensiveness. Our rule for including information in the analysis was whether the target was represented over at least one coastal ecosection. And because we favored data that included a species' specific life stage (i.e., spawning, feeding areas) over data that represented general distributions or observations or modeled data, we were limited by the amount of information included in the analysis. Without a rigorous evaluation through a process similar to the creation of element occurrences, the inclusion of general polygon distribution or observed point locations may not represent the most persistent populations. In addition, marine species data usually does not indicate an association with habitat and is biased to places where positive observations were recorded.

For marine fish we had no fishery-independent survey information, and we did not have any forage fish spawning data for Oregon. Only Pacific Herring spawning data allowed for a comparison across a substantial portion of the ecoregion (British Columbia and Washington). We did collect fisheries-independent trawl data from NOAA, but this did not extend into shallower waters. As noted above, marine technical teams chose not to include general distribution data or local data sets on marine fish (or any other taxa group) because these data were not enough to support selection of priority conservation areas beyond the defined nearshore zone for this assessment. Therefore more intensive survey work needs to be done to sample waters between Mean Lower Low Water and roughly 50 meters. In addition, programs like Essential Fish Habitat (NOAA - EFH) in

Southeast Alaska that sample for juvenile rockfish utilization in estuaries needs to continue and increase in scale.

Marine mammal data were either general distribution areas depicted as polygons, or as random site observations from whale watching vessels. Neither of these data types were included in the analysis, reflecting the general limitation of marine mammal data. Most species are wide ranging and although we have a general sense of their home ranges and migratory corridors we often lack specific site information on feeding areas and other life stages. More work needs to be done to evaluate the use of wide ranging species data in ecoregional assessments and whether models of habitat suitability for these species similar to those done in the terrestrial environment would be useful.

We had spatially explicit data for seabird colonies throughout the region, but shorebird data other than Black Oystercatcher colonies and Western Snowy Plover sites did not represent a specific life stage at the appropriate scale of analysis. Shorebird areas, depicted by large concentration areas using the more explicit area-based estuarine targets, such as tidal mudflats, served as a better surrogate for shorebirds than using the limited occurrence data.

Our largest data gap was for marine invertebrates in the intertidal and subtidal zones. Without a comprehensive, continuous survey effort, we were limited by the places where species were found at distinct locations. These data were used to evaluate the results of the draft portfolio at specific sites, but were not comprehensive enough to use without biasing the analysis. It was therefore difficult to get a sense of abundance of specific vulnerable or threatened species across the region. Although this is a systemic problem for all spatial analyses, it is particularly problematic for sessile invertebrates that may utilize large areas of benthic habitat types. These sparse data reflected neither the best nor the only sites where these species occur. Where there are a very limited number of species observed regionally (i.e., ShoreZone), these data could not be considered indicative of the distribution of the invertebrate communities or be used to track rare species.

2.5 Setting Goals

The analytical tool used in this assessment requires that goals be set for conservation targets. These goals are a device for assembling an efficient conservation portfolio, and they are also first approximations of the necessary conditions for long-term survival of plant communities and ecological systems. Ideally, when setting goals we are attempting to capture ecological variation across the ecoregion and enhance species persistence by spreading the risk of extirpation.

Our objective was to find an efficient number of places to begin addressing conservation in the nearshore; this does not mean that these places capture all that is sufficient to conserve nearshore biodiversity. This approach attempts to answer the question ‘where do we start?’ in evaluating places for nearshore biodiversity, as opposed to ‘how much area is enough?’ to conserve that biodiversity. Given these considerations, we set conservative (low) goals to help the algorithm assemble an efficient portfolio of sites important to multiple targets.

In working with agency partners we agreed that there should be a no net loss of nearshore marine targets. Theoretically goals should therefore be set at 100% of existing occurrences. However, in order to produce an optimized conservation portfolio, we set goals so that the site selection algorithm would have to choose places that capture multiple targets in the fewest possible places. Thus we set goals of between 10 and 50% (see below).

2.5.1 Ecosystem, Shoreline Type and Area-based Estuarine Goals

Goals for coarse filter targets were based on linear meters of shoreline whereas goals for the area-based estuarine targets were based on hectares. ShoreZone data were the most uniform across the ecoregion, providing the best data for describing a portfolio representative of the ecoregion's nearshore habitats. We examined a variety of goal levels for shoreline types ranging from 10 to 40%. Goals were set 10% higher for targets with a biological component (i.e., protected organics/fines) than one without (i.e., exposed rock platform). We initially selected three scenarios, setting goals at 10 and 20%, 20 and 30%, and 30 and 40%. We concluded that the 20 to 30% scenario was appropriate to identify priorities in evaluating the conservation of the diverse coastal environment. Reviewers indicated that the 10 and 20% scenario omitted some critical sites, especially where extensive dikes have been built or invasive species were prevalent but ecological processes were still intact (i.e., adequate fresh and tidal flow regimes in estuaries for juvenile fish rearing habitat). Further, reviewers indicated that the 30 and 40% identified too many sites that were often felt to be low in potential quality. Given that the algorithm attempts to filter a large amount of information into a representative subset, we felt that the 20 and 30% scenario was the appropriate level to test efficiency and overrepresentation of targets within a selection arrangement.

Likewise we examined multiple goals for the coastal ecosystems, also ranging them from 10 to 40% of the target's current extent. Within each scenario we grouped specific targets and gave preference to some by setting their goals 10% higher. Of the seven coastal ecosystems, we set goals 10% higher for: saltmarsh, surfgrass, eelgrass, kelp beds, and rocky intertidal. Goals were 10% lower for dune grasses and algal beds. Marine technical teams determined that these targets were either outside of the intertidal zone (dune grasses in the supratidal) or they were abundant (algal beds) relative to the other targets. However, teams identified these two groups as contributing significantly to the representation of nearshore biodiversity and were therefore retained in the analysis. For the reasons stated above we again chose the 20 and 30% scenario as the optimal setting for site selection. In this way the selection algorithm chose more occurrences of the biologically richest sites to ensure representation of the wider range of species that occupy them. This approach to goal setting attempted to integrate intertidal and shallow subtidal ecosystems with their associated shoreline types.

We conducted a similar procedure for estuaries, establishing the same three goal scenarios and settling on the 20 to 30% range. Similar to the coarse filter shoreline and ecosystem target goals, there was a 10% hike in area-based estuarine targets with a biological component (i.e., wood debris/organic as opposed to sand) as well as preference given to area-based saltmarsh and seagrass targets (goals set 10% lower for dune grasses, aquatic beds, and algal beds).

2.5.2 Species Goals

In setting goals for species targets we considered the relative abundance, distribution, and number of occurrences as well as our confidence in the data. Data sets that were more comprehensive across the ecoregion, recently compiled, or represented a specific life stage (i.e., spawning) as opposed to observational or modeled data, received higher goals. With these factors in mind, we examined various goal scenarios for each taxa group.

We set goal scenarios at 20 and 30%, 30 and 40%, and 40 and 50% for all taxa groups except invertebrates. Forage fish, goals were set 10% higher for Pacific Herring spawning beaches in British Columbia with Relative Importance (RI) values from three to five; all recently surveyed spawning beaches in Washington were given this same goal. All Surf Smelt beaches were also given the same goal level. Steller sea lions represented as rookery sites in British Columbia were given a 10% hike in their goal as opposed to haulout sites. Western Snowy Plover sites were evaluated for their “significant use” during the breeding season and values were 10% higher for locations deemed to have more frequency of utility. For the forage fish, marine mammals, and Plovers the 30 and 40% range was selected for the draft nearshore portfolio. Seabird/shorebird species represented in colonies were all given the same goal, with percentages set at 20, 30 and 40% across scenarios. This was also set for the mussels-barnacles target. The 30% goal of all existing colonies and presence of mussels-barnacles was selected for the draft portfolio.

2.6 Nearshore Marine Suitability Index

The nearshore environment is not easily defined or mapped. It is subject to forces both oceanic and terrestrial, producing ecosystems that are dynamic and "open" in nature. It is not surprising, that coastal areas are affected by human activities in nearby watersheds, the marine environment, and on the shore itself.

When the marine technical teams began designing a nearshore suitability index it was evident that terrestrial, freshwater, and marine impacts had to be considered. The nearshore suitability index refers to factors that either adversely affect the health of an ecosystem (human impacts) or make conserving a particular area less feasible (designation of land use and socio-economic values). Using an index for site selection tends to reduce representation in places where human uses or modifications restrict conservation options. These “costs” may be seen as either more (e.g., lands already in some protected status) or less (e.g., lands devoted to resource extraction) suitable for conservation action.

The nearshore suitability index was characterized around three main categories: a) shoreline impacts, b) adjacent terrestrial, freshwater, and marine factors, and c) management designations across all environments.

2.6.1 Shoreline Impacts

The nearshore has been described as having a high degree of biological productivity, is the part of the marine ecosystem that includes and is most likely influenced by riparian interactions, and is also affected the most from anthropogenic disturbances/interactions (Brennan and Culverwell, in press). Coastal development, a major threat to estuarine and nearshore ecosystem function, alters the physical condition of the shoreline which in turn changes the biological structure and functioning of shoreline habitats (see Shreffler et al. 1994). This affects the use of these habitats by fish, shellfish, birds and other organisms.

In the Pacific Northwest Coast ecoregion some of the most dramatic alteration of the shoreline environment has come from shoreline armoring, or bulkheading. Placing vertical seawalls, riprap, and other coastal structures in the intertidal zone dramatically changes sediment and species composition. In addition, the fish and timber industries heavily utilize the nearshore for transferring logs and growing exotic finfish and shellfish. Logging practices along the coast can lead to significant surficial erosion that result in lost topsoil, siltation and burial of aquatic life. Once the logs are piled in estuaries and embayments they can further damage the coastal environment by impacting the soft bottoms utilized by shellfish and seagrasses . Aquaculture also impacts the nearshore by exposing the environment to high amounts of nitrogen, phosphorus, and fecal matter (see Pew Oceans Commission 2003). Exotic fish that escape their pens can alter native species composition by establishing themselves in surrounding stream systems. Facilities including sewage treatment buildings, pulp mills, and agricultural fertilizer and chemical plants were also considered causes of nearshore species decline and habitat degradation .

All the shoreline impacts were given a relative score. In our scoring system we assumed that finfish tenures or leases for fish farming and log transfer sites have the highest shoreline impacts, followed by coastal structures and facilities. Bulkheads were separated into two categories where they were considered high if the armoring covered at least half of the entire length of a shoreline unit. The shellfish tenures or leases were also broken into two categories, separated by the density of tenures in any coastal area. Most of the sites were determined to be of low density, but a few places in British Columbia contained two or more tenures per shoreline unit. In these cases they were considered to be of high density. Hatcheries were given the lowest relative score because this impact was not considered as detrimental to the nearshore environment. Coastal hatcheries were not considered to be a direct impact to the nearshore, unlike fish farming. We recognize that these scores can be debated and need further examination (e.g., some argue that aquaculture practices are not nearly as detrimental to the marine environment as suggested here, while others view hatcheries as a beneficial factor in increasing fish production and therefore should be removed from the index).

Data sources included shoreline armoring and bulkheading data (ODFW, WDNR, MSRM), fish and shellfish aquaculture sites (WDNR, MSRM, DFO), log transfer sites (MSRM, DFO), coastal hatcheries (MSRM), and industrial/treatment facilities (DLCD, WDNR, MSRM, DFO). Coastal structures have been mapped as point and linear features and attributed to the same spatial data as the shoreline conservation targets. Log transfer sites and tenure data have been mapped as either points or polygons illustrating their general location. The point data were attributed to the shoreline features (e.g., a log transfer site was associated with a linear shoreline segment). Where tenures and log transfer sites were represented as point data they were included in the shoreline impact analysis; where they were represented as areal extents they were included in the adjacency analysis (described in the next section). For both analyses the impact scores were the same. Point locations of coastal facilities were included in the analysis where they were within 500 meters from shore. Due to limitations in the data, hatcheries in Washington and Oregon were not included in the final analysis.

We then calculated a shoreline cost within the nearshore assessment units (400 hectares). If all shoreline impacts occurred in an assessment unit the total shoreline cost would add up to twice the base cost, or 800 (Figure 2.9.10). The shoreline cost was calculated for each assessment unit using:

Shoreline cost = base cost + (base cost * cumulative impact scores)

2.6.2 Adjacent Terrestrial, Freshwater, and Marine Factors

Estuaries have long been recognized as the confluence of a freshwater source and the marine environment (MacKenzie and Moran 2004), but there is a growing amount of attention in the scientific literature regarding the concept of a marine riparian zone across the entire coastal environment (e.g., Desbonnet et al. 1994, Lemieux et al. 2004, Levings and Jamieson 2001, NRC 2002). In their manuscript, Brennan and Culverwell (In press) define the marine riparian zone as "riparian systems located in those areas on or by land bordering a wetland, stream, lake, tidewater, or other body of water that constitute the interface between terrestrial and aquatic ecosystems". The health and integrity of the nearshore ecosystem is significantly influenced by the character of the land adjacent to marine shorelines and the transport mechanisms from both the degree of freshwater flow and tidal flooding.

Commercial and residential development along our coasts is transforming land at an unprecedented rate. Coastal counties, which comprise just 17 percent of the land area nationwide, are now home to more than half of the U.S. population (Pew Oceans Commission 2003). And with another 25 million people living along the coast by 2015 (Beach 2002), our wetlands, estuaries, and other coastal habitats will continue to be strained. As mentioned above, coastal development adjacent to the shoreline is a major threat to the nearshore. Habitat destruction and the decline of coastal water quality resulting from upland development are leading causes of species decline (e.g., Doyle et al. 2001). The natural flow of sediment over land and through waterways is important for sustaining coastal habitats and maintaining beaches (U.S. Commission on Ocean Policy

2004). Too little sediment can lead to habitat decline, damaging wetlands and allowing beaches to wash away over time. However, excess or contaminated sediment can block shipping channels, destroy habitats, poison the food chain, and endanger lives. Navigational dredging, infrastructure projects, farming, forestry, urban development, industrial operations, and many other necessary and beneficial human activities can interfere with natural sediment processes.

Adjacent terrestrial and marine impacts were factored into the nearshore suitability index (Figure 2.9.11). We will incorporate watershed and freshwater characteristics (e.g., drainage area, flow accumulation) in further iterations of the index. Similar to the costs assigned in the terrestrial suitability index, industry and urban areas were assigned a higher score than agriculture and early seral forests. Dredge disposal sites in Oregon and contamination sites in British Columbia were also included in the industrial category. Unlike the terrestrial suitability, however, where road density was calculated as a normalized cost per watershed area, we separated highways and railroads from other roads as higher costs. The length of road was not assigned a cost relative to the size of the assessment unit, but was given an overall weight relative to other land use factors. In addition, we used adjacent marine impacts including the areal extent of finfish and shellfish tenures as well as log transfer sites in estuaries and embayments. These factors are listed under the shoreline impact table.

Unlike the terrestrial suitability index that calculated different cost factors within watershed assessment units, we designed a method of calculating the influence of adjacent land and water conditions that either directly or indirectly affect the shoreline. Adjacent lands were considered to be all watersheds (USGS HUCs, level 6 – the terrestrial assessment units) that directly drain into the coastal zone. The adjacent waters were considered to be all nearshore waters within a 500 meter buffer of the coast.

All factors were combined so that we could calculate cumulative adjacency costs. The analysis of adjacency factors was done in the grid or raster environment, where all data sets were transformed to cell-based or raster data and assigned the relative scores. Data sources included a combination of land use land cover data (Vancouver Island 1:250,000 thematic map by BTM containing 19 classes including 3 seral stages; WDFW contained 10 classes with 3 seral stages; CLAMS or Coastal Landscape Analysis and Modeling Study covered Oregon, containing 4 seral stages; USGS 1:100,000 contained 4 classes, updated by Pacific Meridian Resources) and associated Estuary Plan Book data (DLCD), roads data (compiled by TNC Oregon), dredging disposal sites in the Estuary Plan Book (DLCD), aquatic lands designations from the aquatic ownership data (WDNR), fish and shellfish aquaculture sites (WDNR, MSRM, DFO), log transfer sites (MSRM, DFO), and contamination sites (DFO).

The result of the combined function was to sort all the unique combinations of adjacency cost factors that applied to each cell. We then added all the scores together for every cell and derived a new grid. This grid surface was derived by taking the total score attribute with values greater than zero (value expression). The range of scores across all factors was from .05 to .85.

Since we were interested in the interaction between these costs and their association with the nearshore, we performed a focal function to evaluate all cells in the coastal watersheds as they entered the coastal zone. Focal functions compute an output value based on the values of the input cells within a neighborhood that is centered on the output cell (see Zeiler 1999). All calculations are written to each output cell based on the original input data, creating a new grid surface. The neighborhood uses the values of other locations within a given distance or direction in assigning a value to the output cell. We assigned a neighborhood of roughly 500 meters, or 17 cells wide based on a 30 meter cell size (510 meters), using a circular shaped search zone. We used the focalmean function, which computes the mean of the values in the neighborhood. The parameter 'data' in the formula means that only cell values with a true value were used; cells with no data values were disregarded.

The last step was to populate a single adjacency cost value per assessment unit in the nearshore. We did this for both shoreline (linear) and nearshore (areal) assessment units. We performed a zonal function to assign a mean value to each assessment unit. Zonal functions compute statistics for a value grid (the focalmean grid) by zones defined in a zone grid (assessment unit grid). The output was a table of statistics for each assessment unit. We used the mean of all focal mean values in each assessment unit, and applied a similar equation as the shoreline cost:

Adjacency cost = base cost + (base cost * mean of adjacency impact scores)

2.6.3 Management Designations across all Environments

We recognize that assigning relative values to management designations in the marine environment according to their level of protection is more difficult than on land. This difficulty arises because there are often multiple factors to consider including what is being protected, what portion of the marine environment is actually within designated boundaries, and what uses are allowable. For instance, the Olympic Coast National Marine Sanctuary (OCNMS or sanctuary) was designated in 1994 as part of the federal National Marine Sanctuary System. The area was recognized for its extraordinary beauty and rich biological diversity, as a marine area deserving of enhanced protection and preservation (OCNMS Advisory Council 2003). OCNMS covers approximately 8550 square kilometers of the outer coast of Washington, stretching north from the Copalis River around Cape Flattery to Koitlah Point, approximately 4 nautical miles into the Strait of Juan de Fuca. OCNMS was established as a multiple use marine protected area, with mandates for resource protection, research, and education, but with relatively few restrictions on human activities. Activities prohibited by sanctuary regulations include overflights below 2000 feet within 1 nautical mile of the coast or national wildlife refuge islands, oil exploration and drilling, extraction of ocean minerals, alteration of the seafloor with the exception of traditional fishing practices, and discharge and deposit of materials. The marine conservation working group's final report for the sanctuary (OCNMS Advisory Council 2003) recognizes that although existing regulations do provide a level of protection to meet the sanctuary's mission of ecosystem-wide conservation of ecological and historic resources, activities such as gathering of intertidal resources and bottom trawling continue to occur at levels that are poorly documented or in ways that might contribute to habitat degradation. This, along with state and tribal jurisdiction and rights within the sanctuary, further complicate an assessment of marine protection.

Among the three major categories of the suitability index, we scored the shoreline impacts highest followed by adjacency factors and management designations. We used the work of the Gap Analysis Program as a baseline for scoring management designations (see Cassidy et al. 1997, Kagan et al. 1999). We assumed that protected and natural area designations receive less human impact and are managed for biodiversity relative to areas designated as pubic resources or private lands (Figure 2.9.12). The protected areas category included marine protected areas in British Columbia, the National Marine Sanctuary, National Wildlife Refuges, National Estuarine Research Reserves, National Parks, Wilderness areas, and Nature Conservancy preserves. The natural areas category included state, provincial, and county parks, marine gardens, and research reserves. Other public lands designated for multiple-use were given a higher score, but ranked lower than all private lands. The public lands category included National Forest Service, Bureau of Land Management, British Columbia Crown Lands, and designated public tidelands and bedlands. We assumed that private industrial lands, commercial industry, or areas projected for industrial development represented the highest potential impact. Private lands including tribal reservations, oyster tracts, and urban areas were given a slightly lower score (private lands/urban). These scores were utilized in the initial analysis of management designations and will be refined as we further examine levels of protection and impact resulting from these designations.

We conducted the same methods utilizing focal and zonal functions to calculate management costs. Data sources included a combination of land use/land cover data (Vancouver Island 1:250,000 thematic map by BTM containing 19 classes including 3 seral stages; WDFW contained 10 classes with 3 seral stages; CLAMS or Coastal Landscape Analysis and Modeling Study covered Oregon, containing 4 seral stages; USGS 1:100,000 contained 4 classes, updated by Pacific Meridian Resources),associated Estuary Plan Book data (DLCD), aquatic lands management designations from the aquatic ownership data (WDNR), aquatic lands designated as protected or reserve areas from the aquatic ownership data (WDNR), marine protected areas (DFO), and log transfer sites (MSRM, DFO).

All unique combinations of management cost factors were then summed across all cells. The attribute for all summed factors was then selected to produce a cumulative grid. This grid surface was derived by taking the total score attribute for values greater than zero (value expression). The range of total weights across all factors was from .01 to .60.

In a similar fashion to computing the adjacency costs, we assigned a neighborhood of roughly 500 meters, or 17 cells wide based on a 30 meter cell size (510 meters), using a circular shaped search zone. We used the focalmean function, which computes the mean of the values in the neighborhood. The parameter 'data' in the formula means that only cell values with a true value were used; cells with no data values were disregarded.

The last step was to populate a single management cost value per shore and nearshore assessment unit. We took the mean value of all focalmean values in each assessment unit and applied the same equation as the shoreline and adjacency costs:

Management cost = base cost + (base cost * mean of human impact scores)

The final step of the suitability analysis was to combine shoreline, adjacency, and management costs to produce an overall cost per assessment unit. Up to this point we had constructed standalone costs to test the sensitivity of site selection for each category. After a preliminary assessment, we compiled all three categories in a single index. The formula to compile the overall index was:

Overall suitability index = base costs + shoreline costs + adjacency costs + management costs

The range of the suitability index was from 400 to 752 for the nearshore units. Initially we designed three scenarios for the overall index. The first assigned all assessment units equally using either the shoreline or nearshore base cost. The second scenario is illustrated in the explanation of methods and scores above. We also conducted a third scenario that increased the range of scores across all factors to see how site selection would be affected. After testing these scenarios we determined that the scores and process described above validated conditions on the ground and in the water, and that site selection was more accurately represented using the second scenario (Figure 2.9.13). This scenario best supported the optimization of the least area needed to meet the conservation goals for all targets.

2.7 Nearshore Marine Portfolio

2.7.1 Planning Units

In the development of the nearshore marine portfolio we utilized a decision support tool called MARXAN, an optimal reserve site selection algorithm (see Andelman 1999, Possingham et al. 2000). Similar to SITEs, MARXAN helps create an efficient conservation portfolio by minimizing the total area selected while meeting the assigned conservation goals. MARXAN was developed as a modified and updated version of SITEs to meet the needs of the Great Barrier Reef Marine Planning Authority (GBRMPA) in their rezoning plans and is currently the most widely used decision support tool for marine reserve system design in the world (http://www.ecology.uq.edu.au/index.html?page=27710).

Like SITEs, MARXAN requires that the ecoregion be divided into a set of candidate sites, or assessment units, that completely fill the region and utilizes a simulated annealing algorithm to evaluate alternative site selection scenarios. The algorithm's objective function is a nonlinear combination of the total area and the boundary length of perimeter of the site selection output (Leslie et al. 2003). There is never just one "optimal" solution (e.g., the definitive set of conservation areas) in regional planning, but it is possible to identify those areas that are both essential and representative as part of an ecological assessment. However, siting algorithms provide a context for objective representation that is both measurable and spatially explicit.

Marine technical teams designed an analytical framework to evaluate the different output from MARXAN and test those analyses within a structured review process (Figure 2.9.14). Initial sensitivity analyses (Tiers 1 - 3) were conducted to test the number and classification of conservation targets included in the analysis, variations in the suitability index, and spatial formats of Aus (need to define AU once). Tier 4 was designed to vary conservation goals across all targets and the amount of shared boundary (the clumping or boundary modifier) between AUs. Tier 5 incorporated expert review recommendations into the various algorithmic solutions to complete the final draft nearshore marine portfolio. This portfolio was then integrated with terrestrial and freshwater portfolios to form land/sea conservation areas. This framework also takes into account spatially explicit threats, both land and marine-based factors that affect the coastal environment.

The design and selection of the appropriate assessment unit is heavily debated within and among conservation planners (Ferdaña 2005). Choosing the spatial configuration and size are the two main debatable components. Tier 2 was designed to examine various assessment unit configurations. These included two natural units of analysis, shorelines and estuaries, and one abstract unit, grids. Natural units are generally of variable size that fit within ecological boundaries. Examples are watersheds determined by drainage area and shoreline segments or reaches determined by the length of a dominant beach substrate. Abstract units are generally equally sized areas that arbitrarily fall across the land and/or seascape. Examples are grids or hexagons. All the different units have their advantages and disadvantages, and therefore initially using the shoreline segments, estuary polygons, and grid units helped test the algorithm in identifying highly valued areas regardless of spatial format.

Shoreline assessment units are represented as linear features defined by dominant beach types. These units were the same spatial data that represented the shoreline targets as well as the linear intertidal/shallow subtidal vegetated habitats. They are highly variable in length, providing the most spatially explicit unit of analysis. The algorithm used this unit to identify shorelines that were either tens or thousands of meters long. We initially used this spatial format but given the variable length we decided not to use this unit in representing the nearshore portfolio. This unit, however, formed the basis for the nearshore component of the Puget Trough/Georgia Basin ecoregional assessment (Floberg et al. 2004). For a detailed examination and discussion on shoreline assessment units versus abstract units see Ferdaña (2005).

We also constructed estuary assessment units from various estuary data across the ecoregion. We used a NOAA salinity data set (2001) to separate the larger estuaries into saltwater, mixing, tidal fresh zones, and islands within estuaries in Oregon and

Washington. Estuary assessment units provided a polygonal format for evaluating benthic substrate and habitat types as well as species information that was more difficult to capture with linear shorelines. This represented an areal unit for analyzing estuarine ecosystems and broad embayments, and was associated with HUC watershed assessment units (USGS watershed system) in the terrestrial analysis. This was done to find efficiencies in site selection between watersheds and their adjoining estuaries. We compared this analysis with the nearshore and estuary grid selection (described below) prior to the expert review process. We used both estuarine selection methods in building an integrated portfolio with terrestrial site selection.

An abstract assessment unit such as grids has an advantage in that different sizes can be nested. This is an important factor when considering the spatial resolution or nearshore data as opposed to offshore information. We used this approach to assemble nearshore and offshore grids, which we term a nested grid assessment unit (Figure 2.9.15). The nearshore grid size was set after evaluating all input data scales and resolutions. This grid accommodated all shoreline and estuarine information as well as species data in the nearshore. The nearshore grid size was 400 hectares. The offshore grid unit was four times larger (1,600 hectares) than the nearshore unit and nested within the larger one. We used the approximate depth of the photic zone, or the depth of macrophytes, to determine the transition between nearshore and offshore grids. We set this transition at the 40-meter depth. Although we have populated only the nearshore grids in this analysis, we plan to do an analysis of the entire offshore area as part of ongoing research in the Pacific Northwest Coast marine environment.

2.7.2 Decision Support System

Using the nearshore grid assessment unit, we populated them with spatial data on targets and the suitability index. With this information we created MARXAN input files and ran scenarios. A scenario in MARXAN corresponds to a set of input parameters, including the desired number of solutions or runs and the amount of clumping of AUs. One of the main outputs, called "summed irreplaceability" or “summed solution” in SITEs, adds all of the solutions from a scenario together. This output keeps track of how often each assessment unit was selected in a given scenario. This information is a useful way to explore the irreplaceability of sites (Leslie et al. 2003, Warman et al. 2004). A "sum of summed solution,” or multiple scenarios added together, has also been referred to as an irreplaceability analysis. Stacking these scenarios allowed us to vary specific MARXAN parameters and track how each assessment unit was selected. We used these scenarios as part of Tier 4 of our analytical framework

The nearshore marine portion of the ecoregion followed the methods of Rumsey et al. (2004) to explore this analysis and assist in the construction of priority areas. We varied the goals across targets (goal ranges of 10 - 30%, 20 - 40%, and 30 - 50%) and amount of clumping or boundary modifier (.01, .05, .125) to produce an irreplaceability map (Figure 2.9.16). We used the output of this analysis for visualizing the range of values (1 – 900, or 9 scenarios run 100 times each) across assessment units that were chosen in at least one solution. Displaying the range of values in equal intervals or solution ranges informed experts of the variability of biodiversity values across the ecoregion (Figure 2.9.17). The lowest 20% range illustrated the spatial variability of 1,317 units chosen 1 to 181 times (64%) across a total of 2,042 units. The highest 20% range identified the most irreplaceable sites, here 177 units chosen 721 to 900 times (9%). This was the range we identified as the core group of units needed in the final portfolio; the other solution ranges helped inform the value of assessment units but were not necessarily chosen for the final. It is generally thought that assessment units chosen more than 50% of the time are essential for efficiently meeting biodiversity goals (from Hugh Possingham's explanation of MARXAN - http://www.ecology.uq.edu.au/index.html?page=20882). Sites that are rarely selected can be ignored. We have chosen to use "irreplaceability" in describing this analysis because of the published literature that supports this term as well as terminology described within the MARXAN decision support tool (see Ball and Possingham 2000, Leslie et al. 2003, McDonnell et al. 2002, Warman et al. 2004). This concept is inspired by, but different from, Bob Pressey's notion of irreplaceability (Pressey et al. 1994).

The other main MARXAN output is called "best solution" which is the most optimal run in the scenario. The scenario for the best solution here used the .01 clumping factor and the 20 - 40% goal range. This single solution was compared to both the irreplaceability map and expert review by marine technical teams in constructing the initial nearshore portfolio. We utilized the summed solution output to examine how many times a single assessment unit chosen in the best solution across the irreplaceability range (Figure 2.9.18). Here we can see that of the 425 units in the best solution, 167 (39%) captured the highest 20% of the irreplaceability range. On the other end of the scale the 36 units (8%) in the lowest 20% range and the 66 units (16%) in the next highest 20% range were considered replaceable. Doing this evaluation for the expert review process helped focus our attention on the highest priority conservation sites in the nearshore environment.

2.7.3 Expert Review

In approaching the expert review process we illustrated the comparison of site selection between grid assessment units and the estuary selection conducted within the terrestrial/freshwater analysis. Overall, there was considerable agreement between the terrestrial/freshwater analysis and the marine analysis, particularly with regards to selected estuaries. The grid units often provided a more spatially explicit selection of estuaries and, where appropriate, delineated specific sections within larger estuaries. These methods were integrated into a single layer of estuarine portfolio sites for the final version.

Marine portfolio reviews of the draft marine portfolio took place in Washington and Oregon. In addition to the portfolio reviews there were several meetings held in each state or province during the data collection phase of the marine assessment. These meetings not only provided data resources but they also reviewed target lists, methods, and recommended specific coastal locations within the ecoregion. In addition, representatives of the three marine technical teams assisted in the construction of the analytical framework.

2.7.4 Integrated Conservation Portfolio

The conservation portfolio is comprised of terrestrial and freshwater lands and waters as well as nearshore intertidal and shallow subtidal marine lands. Most conservation sites comprised aquatic, terrestrial and in some cases marine lands totaling 3,636,996 hectares (8,987,381 acres) over the entire ecoregion. There were four conservation sites, totaling 1047 ha, that were identified solely for their aquatic conservation targets and there were 20 conservation sites, totaling 13,970 ha, identified solely for their marine conservation targets. The portfolio included a considerable area of land already managed for conservation that covered 898,634 ha. These conserved sites often formed core areas within larger watersheds identified in the assessment.

The conservation portfolio includes 164 priority conservation areas that vary widely in size. Not surprisingly the largest sites, Olympic National Park (420,442 ha) and Strathcona (320,906 ha), are dominated by large publicly protected landscape sites. Many of the smaller priority conservation areas, such as Myrtle Island RNA (9 ha) and Copalis Rocks NWR (12 ha), are also publicly protected areas. Many of the smallest priority conservation areas were selected to conserve established protected areas and they often contain only a few conservation targets. This is in contrast with the largest priority conservation areas that were complete landscapes, often comprising several watersheds and including a plethora of target species as well as representative terrestrial and aquatic systems. Both types of conservation areas are critical to protecting the representative biodiversity in the ecoregion but their roles in such conservation may differ considerably. Smaller sites may contain isolated occurrences of rare species or special habitats that are important for maintaining genetic diversity in a regional perspective. Larger sites often contain the best examples of functional land and seascapes that may contribute to biodiversity conservation by maintaining ecological processes that are essential for ecosystem resiliency. Larger sites are also better suited to adapting to climate change and other large scale events that affect biodiversity.

The conservation portfolio is fairly evenly divided between the states and province with a slightly larger percentage of it located in British Columbia (Vancouver Island) and Washington. These minor differences can be attributed to the two largest portfolio sites being located in these regions (Figure 2.9.19).

Length of shoreline is also a good measure for evaluating and summarizing what is captured in the conservation portfolio. These numbers reflect the total length of shoreline in the ecoregion (including man-made and undefined shoreline units) and their distribution across the geopolitical region (Figure 2.9.20).

Another perspective of the conservation portfolio for the Pacific Northwest Coast is with regards to how it is distributed among the terrestrial ecological sections of the ecoregion (Figure 2.9.21). Large, publicly protected sites pushed the conservation portfolio percent of several sections to over 50%. Similarly, the distribution of shoreline across marine ecosections can be displayed (Figure 2.9.22).

2.7.5 Conservation Target Assessment

One of the measures of the effectiveness of the conservation portfolio is the assessment of how well the conservation targets met their assigned conservation goals. The portfolio portrayed in Figure 2.9.23 does a good job of meeting the assigned goals for fine filter and coarse filter targets. Results for groups of targets are summarized in Figure 2.9.24. A complete assessment of how well each conservation target met its goals within each of the ecoregional sections is included in Vander Schaaf et al. 2006.

Nearshore marine conservation targets did very well meeting their MARXAN goals with only a few targets falling short. Coastal shoreline target goals (84% met) did least well among marine target groups but a closer look at this group of targets shows that most of these targets are represented within 80% of their stated conservation goals in the portfolio. All marine fine filter targets met their goals except seabirds as there was no difference between conservation and MARXAN goals. Of all the target groups used in the ecoregional assessment, spatially explicit and comprehensive information on marine species represented the largest data gaps.

It should be noted that expert review and integration processes change SITEs or MARXAN selection output. Often this results in an inflated portfolio area in an attempt to align terrestrial and marine priorities. Where the algorithms are set to optimize site selection by including the most spatially efficient portfolio, expert review and integration methods tend to focus on ecological accuracy across the land/sea interface. The result of this ecological alignment is an increase in overall portfolio size and an overrepresentation (target met by > 130%) of many conservation targets (see Leslie et. al 2003 and Ferdaña 2005).

2.8 Marine Managed Areas

Along the Pacific Northwest Coast lie a number of marine managed areas that offer varying levels of protection to the marine environment. These areas are relatively recent additions to the protected area network and generally offer less than full protection of the biodiversity they contain, as fishing is often not prohibited within designated marine managed areas. Nevertheless, these areas offer critical protection to at least some of the biological attributes and habitats present, and are therefore important in coastal and nearshore conservation. For a more thorough review of marine managed areas off of the U.S. West Coast see Nowlis (2004).

The sites vary considerably in their size with the largest area being the Olympic Coast National Marine Sanctuary, which covers 857,000 ha (3,310 square miles) of ocean off the Olympic coast of Washington. The Sanctuary is managed by NOAA but it borders lands owned by the Makah Nation as well as other lands managed by the National Park Service, State of Washington and private landowners. Fishing is regulated but not prohibited within the Sanctuary. The Sanctuary is the largest marine managed area on land or sea within the ecoregion.

Many marine managed areas are located on Vancouver Island where the government has designated Ecological Reserves, Provincial Parks, a Wildlife Management Area, Pacific Rim National Park and several other designations that involve marine resources. There are at least 30 Provincial Parks that provide some protection for marine or estuarine resources on Vancouver Island.

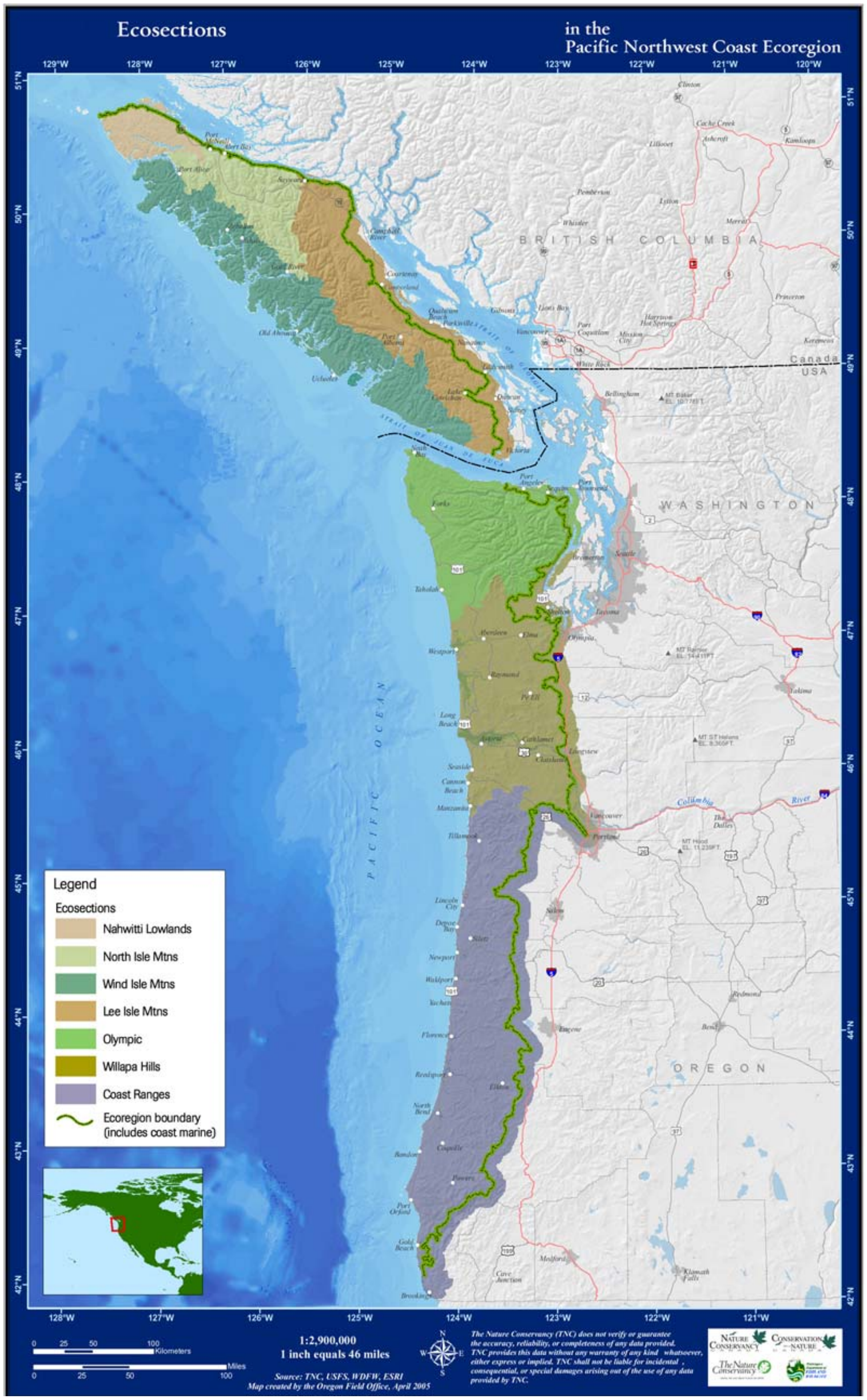
Washington State has a number of marine or coastal protected areas that are located in coastal estuaries or on the outer coast. There are six coastal National Wildlife Refuges (NWR) including two refuges that are offshore, Quillayute Needles NWR and Flattery Rocks NWR. There are also six Natural Area Preserves managed by the Washington State Department of Natural Resources that include marine or estuarine habitats. Finally, there are other state designations such as Seashore Conservation Areas that protect portions of the marine environment.

The Oregon coast has six National Wildlife Refuges including one comprised of every offshore island on the Oregon coast (the Oregon Islands NWR), although protection for subtidal habitats on the islands is limited. Oregon also has the only National Estuary Research Reserve in the ecoregion at South Slough, and two National Estuary Program sites, Tillamook and the Lower Columbia (which falls within Washington as well). Land use zoning offers varying amounts of protection to all of Oregon's estuaries, with nearly half the estuaries protected in their natural state, limiting commercial development and dredging. There are 20 coastal sites and offshore reefs regulated by ODFW with designations that offer seasonal closure and protection from some activities, such as collecting marine organisms for non-research purposes.

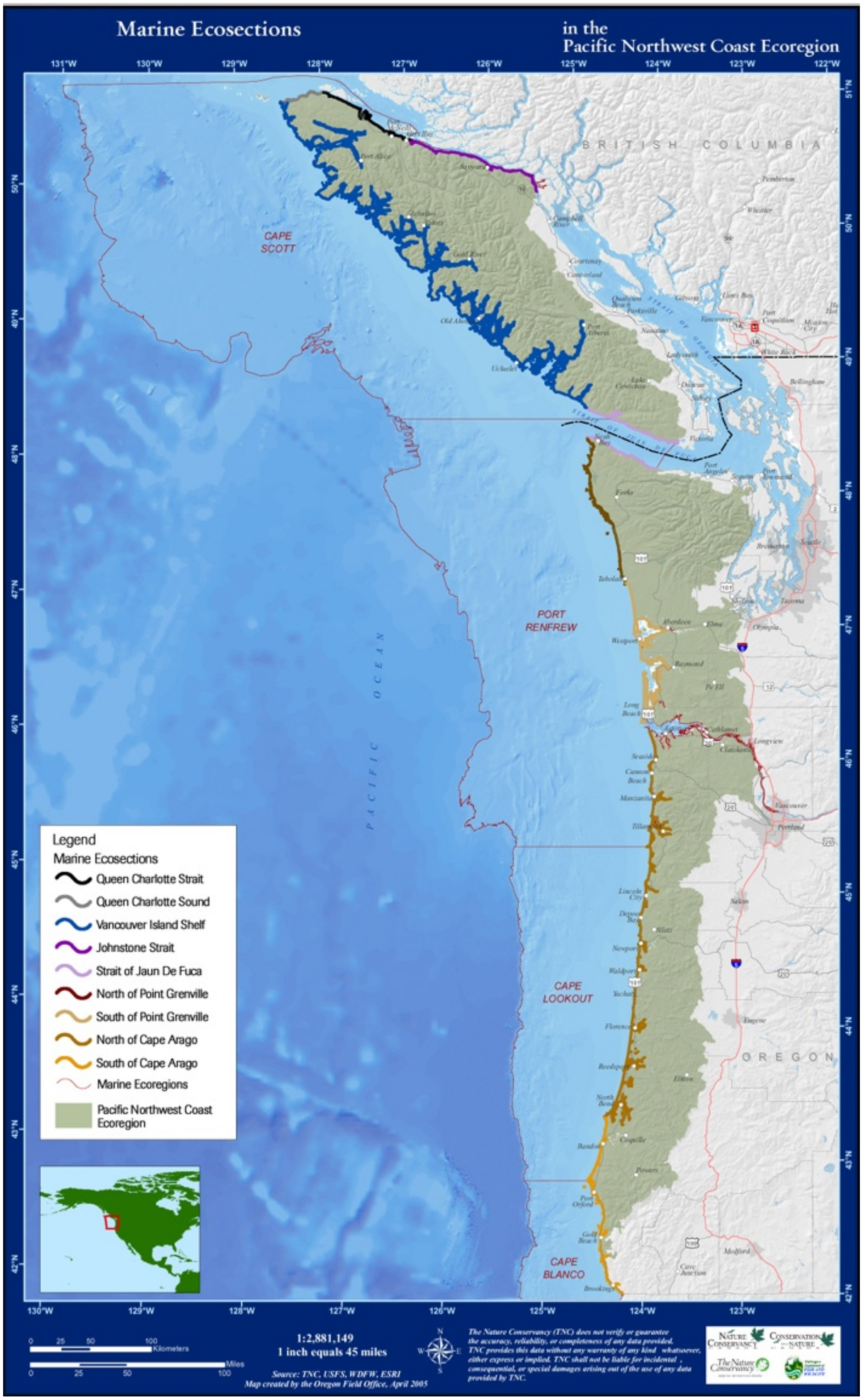
We are currently conducting a gap analysis to identify biodiversity (i.e., species, ecosystems and ecological processes) not adequately conserved within the protected area network or through other effective and long-term conservation measures. Gap analyses have been developed over the past 15 years in response to recognition that protected area systems of all types and in all parts of the world currently do not fully protect biodiversity (Scott et al. 1993). For an in depth treatment of the gap analysis approach see Dudley and Parrish (2005).

2.9 Figures

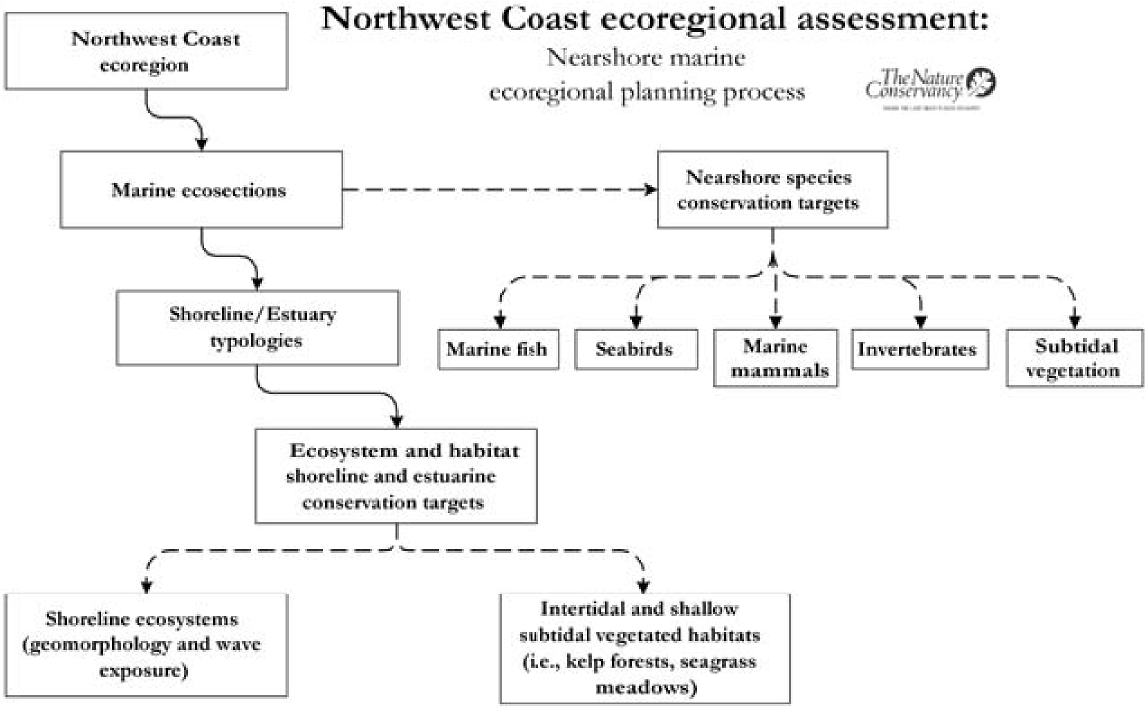
2.9.1 Pacific Northwest Coast ecoregion



2.9.2 Coastal stratification units



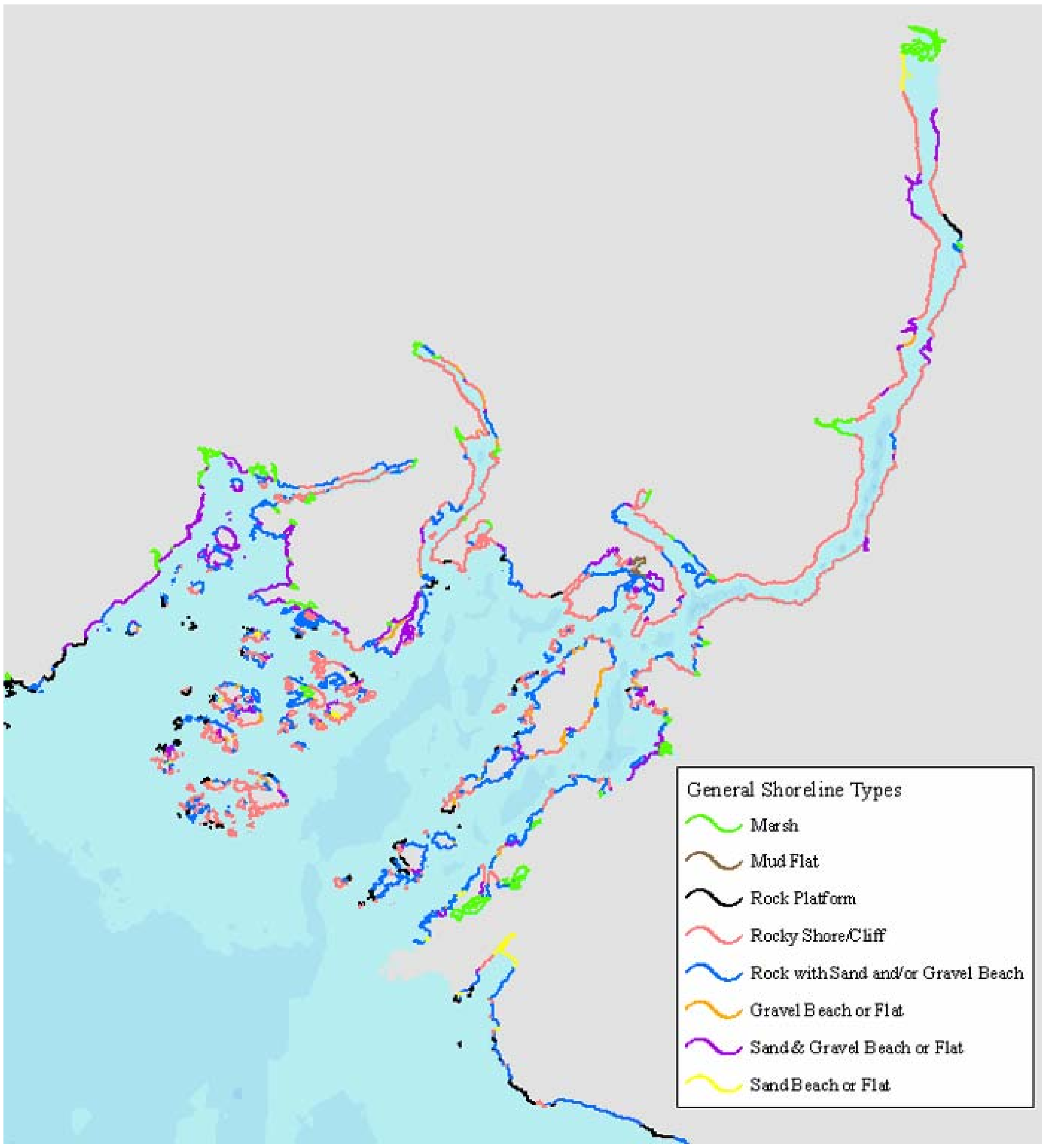
2.9.3 Process undergone to categorize conservation targets



2.9.4 Coastal ecosystems as conservation targets

Target	Area (ha) or Length (m)
Kelp low persistence (WA)	2,308 (ha)
Kelp medium persistence (WA)	1,064 (ha)
Kelp high persistence (WA)	1,121 (ha)
Kelp (OR, BC)	19,479 (ha)
Algal beds	11,282 (ha)
Aquatic bed	657 (ha)
Dune grass	589 (ha)
Eelgrass	1,480 (ha)
Saltmarsh	10,557 (ha)
Seagrass	32,891 (ha)
Algal beds	3,506,778 (m)
Dune grass	797,238 (m)
Rocky intertidal	999,624 (m)
Kelp	1,511,711 (m)
Saltmarsh	2,021,687 (m)
Surfgrass	1,233,673 (m)
Eelgrass	1,190,544 (m)

2.9.5 General Shoreline Types in Barkley Sound, west coast Vancouver Island, British Columbia, Canada.



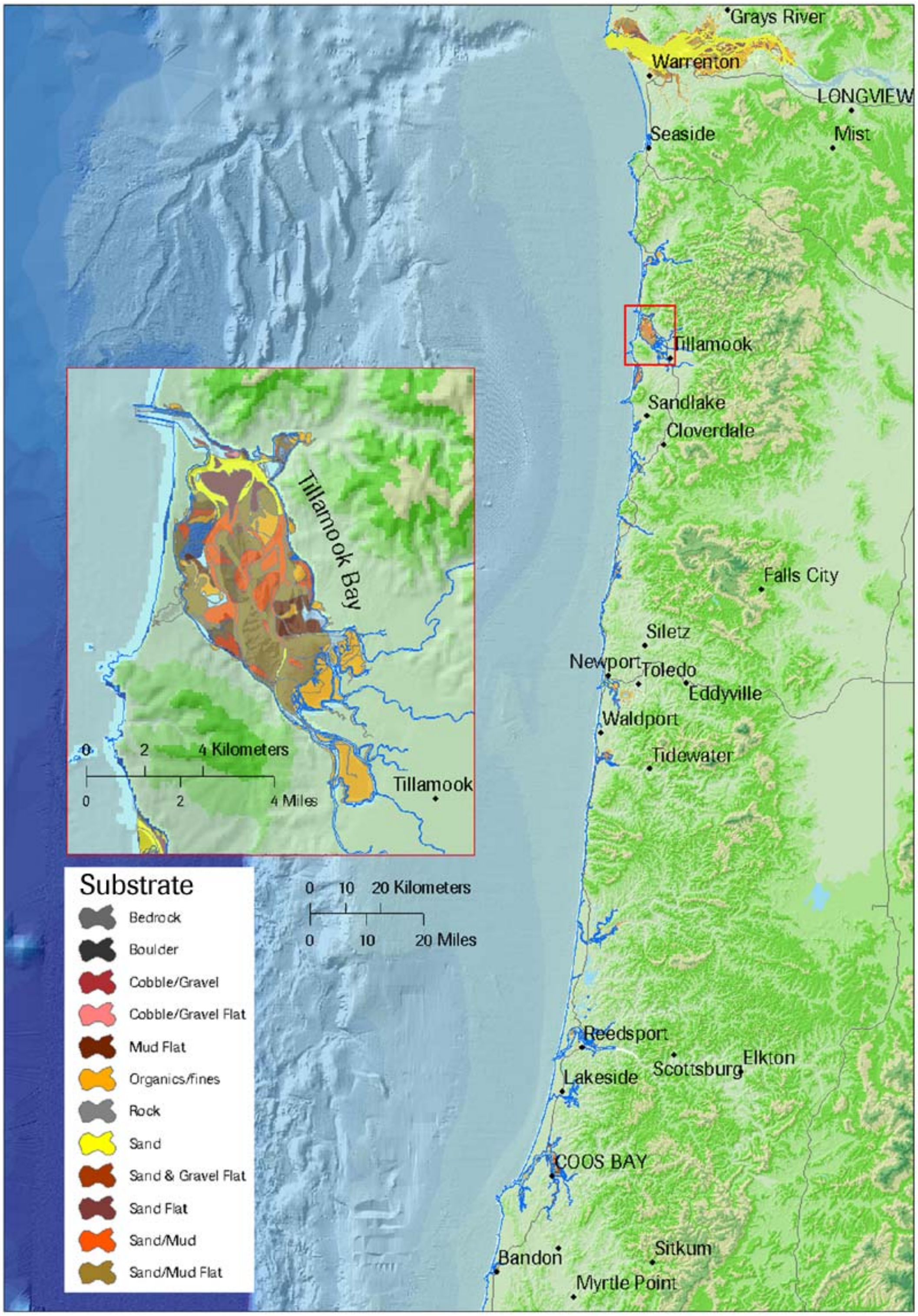
2.9.6 Shoreline types as conservation targets

Lengths in meters Landform	Exposure					
	Very exposed	*Exposed to semi-exposed*	*Semi-protected to protected*	*Very protected*	*Undefined*	**Total**
Channel	0	0	10,705	0	2,296	13,001
Organics/fines	0	485,792	921,277	103,162	199,683	1,709,914
Gravel Beach	65,034	67,647	123,035	11,500	21,886	289,102
Gravel Flat	0	6,876	28,261	0	4,103	39,241
High Tide Lagoon	0	444	9,110	0	0	9,554
Mud Flat	0	2,914	30,566	9,378	20,827	63,685
Rock Platform	23,165	328,567	18,292	0	126,133	496,157
Rock with Gravel Beach	10,731	219,792	661,417	0	105,994	997,935
Rock with Sand & Gravel Beach	2,791	454,827	457,772	0	99,547	1,014,937
Rock with Sand Beach	12,023	192,714	66,860	0	10,768	282,364
Rocky Shore/Cliff	0	502,038	806,639	0	393,449	1,702,126
Sand & Gravel Beach	120,979	78,392	228,796	7,526	38,685	474,379
Sand & Gravel Flat	0	25,272	262,303	0	89,862	377,437
Sand Beach	293,473	204,143	70,027	8,148	68,242	644,033
Sand Flat	103,328	86,531	146,369	3,972	11,234	351,434
Total	631,524	2,655,949	3,841,431	143,686	1,192,710	**8,465,299**

2.9.7 Estuarine types as conservation targets

Substrate	Area (ha)
Bedrock	65.3
Boulder	133.6
Cobble/Gravel	182.6
Cobble/Gravel Flat	199.5
Flat	931.6
Mud	516.6
Mud Flat	30,562.8
Organics/fines	18,325.0
Rock	71.4
Sand	26,590.8
Sand & Gravel Flat	716.9
Sand Flat	10,229.4
Sand/Mud	4,167.1
Sand/Mud Flat	8,501.8
Shell	16.9
Unconsolidated	597.7
Undefined Beach/Bar	22.1
Wood Debris/Organic	25.5
Grand Total	101,856.4

2.9.8 Estuarine substrate types in Tillamook Bay, Oregon



2.9.9 Marine species as conservation targets

Target	Taxa	Currency	Amount
Mussels and barnacles	Invertebrate	Meters	1,124,485
Smelt spawn	Marine Fish	Meters	42,347
Herring spawn high significance	Marine Fish	Meters	281,119
Herring spawn medium significance	Marine Fish	Meters	751,722
Steller sealion haulout	Marine Mammal	Occurrences	41
Steller sealion rookery	Marine Mammal	Occurrences	4
Black Oystercatcher	Seabird	Occurrences	357
Brandt's Cormorant	Seabird	Occurrences	101
Cassin's Auklet	Seabird	Occurrences	18
Caspian Tern	Seabird	Occurrences	4
Common Murre	Seabird	Occurrences	101
Double-crested Cormorant	Seabird	Occurrences	50
Fork-tailed Storm Petrel	Seabird	Occurrences	14
Leach's Storm Petrel	Seabird	Occurrences	36
Pelagic Cormorant	Seabird	Occurrences	316
Pigeon Guillemot	Seabird	Occurrences	386
Rhinoceros Auklet	Seabird	Occurrences	16
Tufted Puffin	Seabird	Occurrences	94
Western Snowy Plover nesting - high significance	Shorebird	Hectares	1,295
Western Snowy Plover nesting - medium significance	Shorebird	Hectares	7,208
Western Snowy Plover nesting points	Shorebird	Occurrences	8

2.9.10 Shoreline impacts, relative scores and associated costs

Shoreline impacts	Impact scores	Base cost	Shoreline costs
Bulkhead high	0.25	400	100
Bulkhead low	0.15	400	60
Finfish tenure	0.35	400	140
Shellfish tenure high	0.3	400	120
Shellfish tenure low	0.2	400	80
Hatcheries	0.15	400	60
Facilities	0.25	400	100
Log transfer sites	0.35	400	140

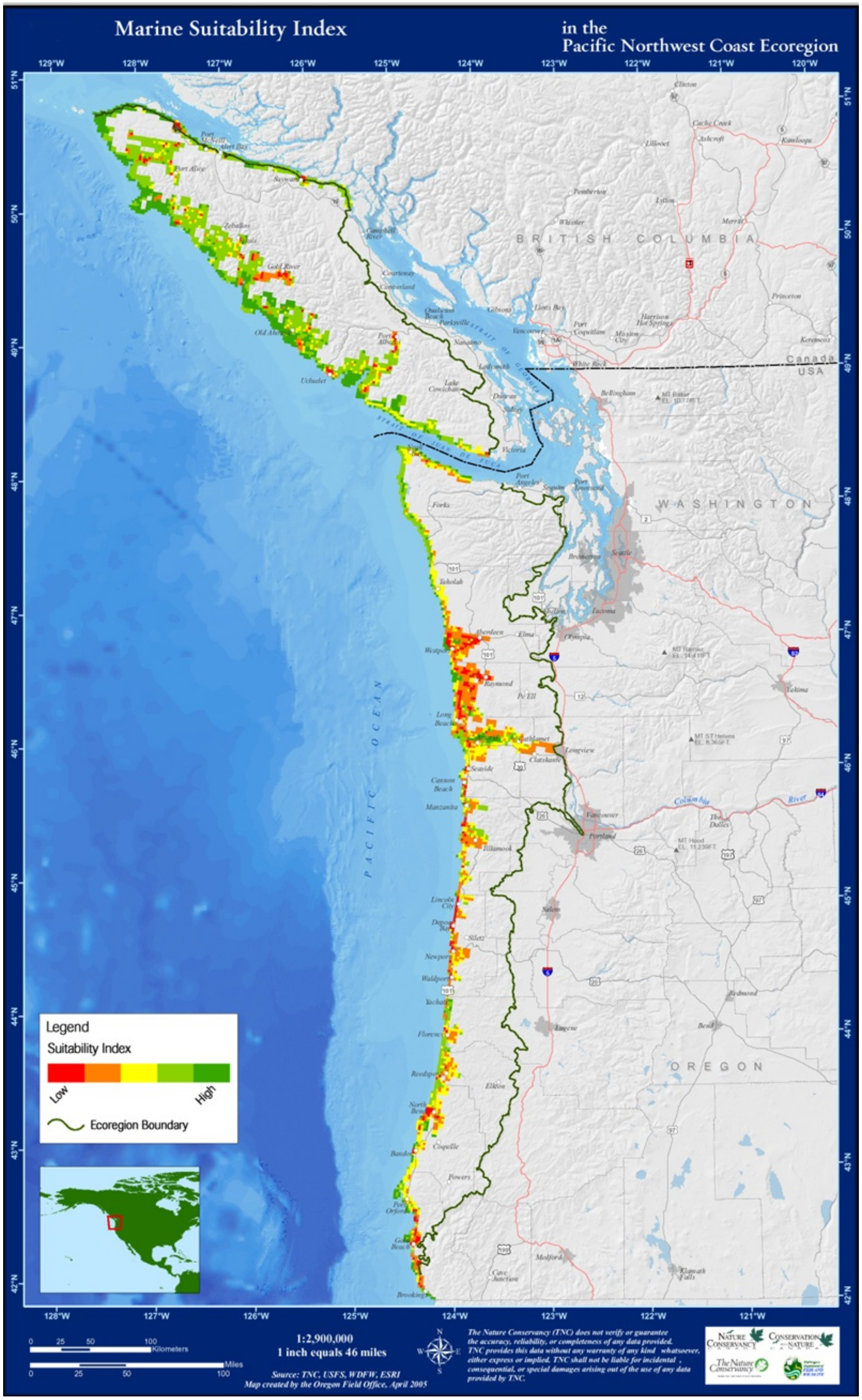
2.9.11 Adjacency factors, individual scores, and associated costs

Adjacency or land use impacts	Impact scores	Base cost	Adjacency costs
Early Seral	0.05	400	20
Agriculture	0.15	400	60
Urban	0.25	400	100
Industrial	0.35	400	140
Roads/Secondary	0.15	400	60
Highways/Railroad	0.25	400	100

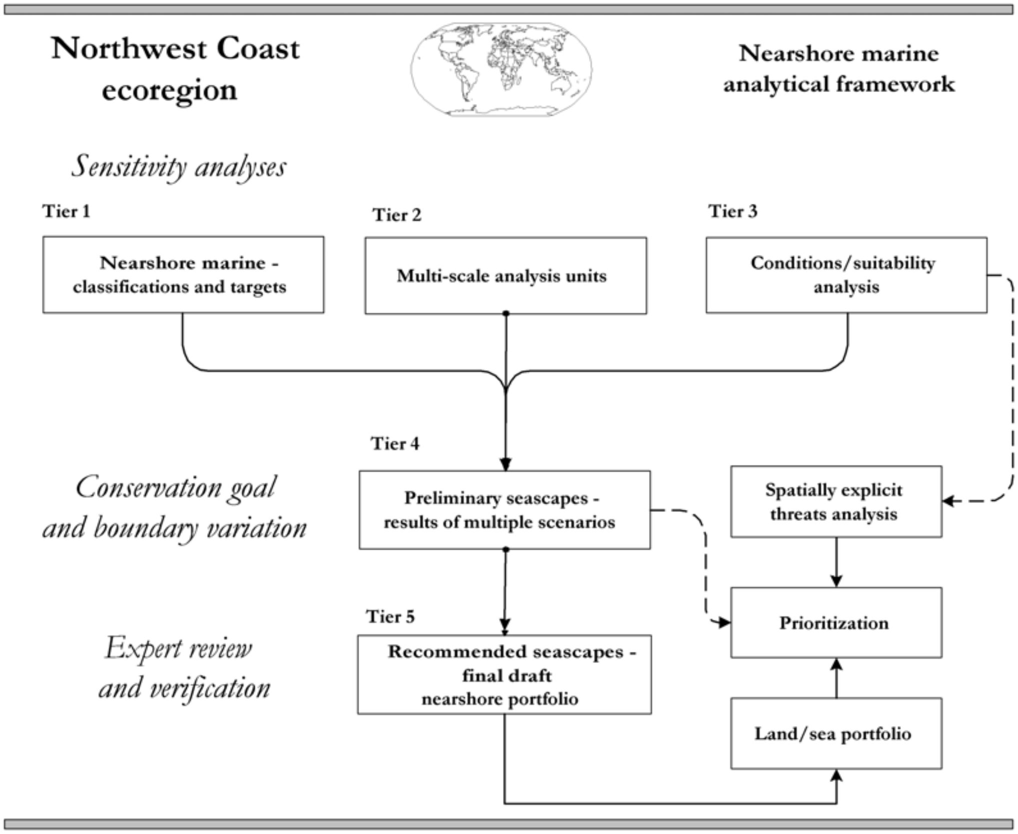
2.9.12 Management factors, individual scores, and associated costs

Management factors	Impact scores	Base cost	Management costs
Protected areas	0.01	400	4
Natural areas	0.05	400	20
Public\multiple use areas	0.15	400	60
Private lands\urban	0.25	400	100
Private lands\industrial	0.35	400	140

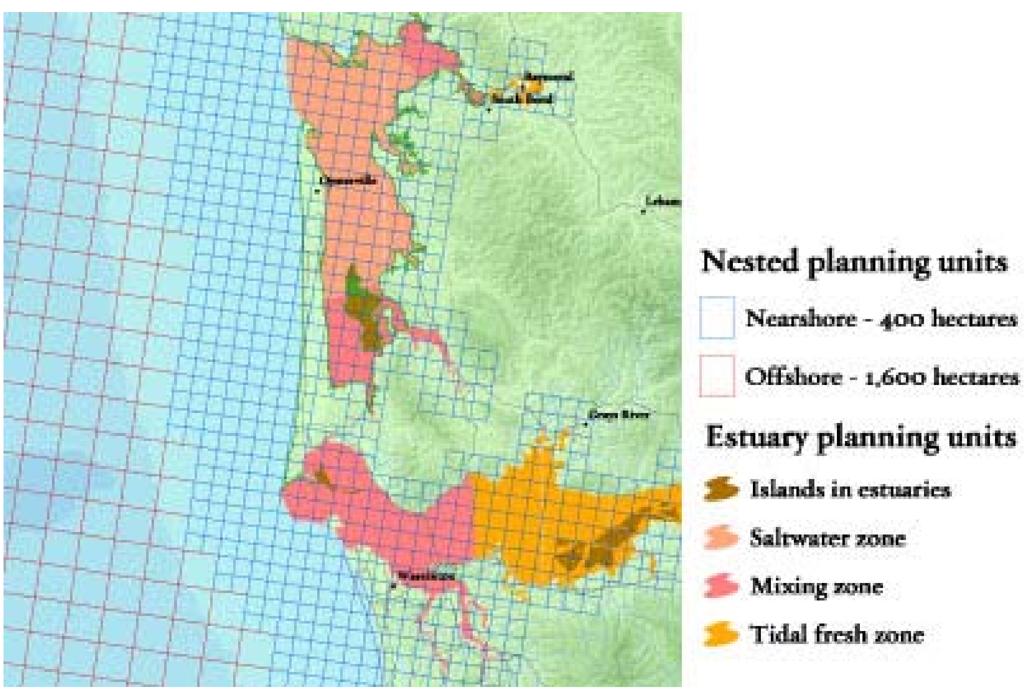
2.9.13 Nearshore marine suitability index



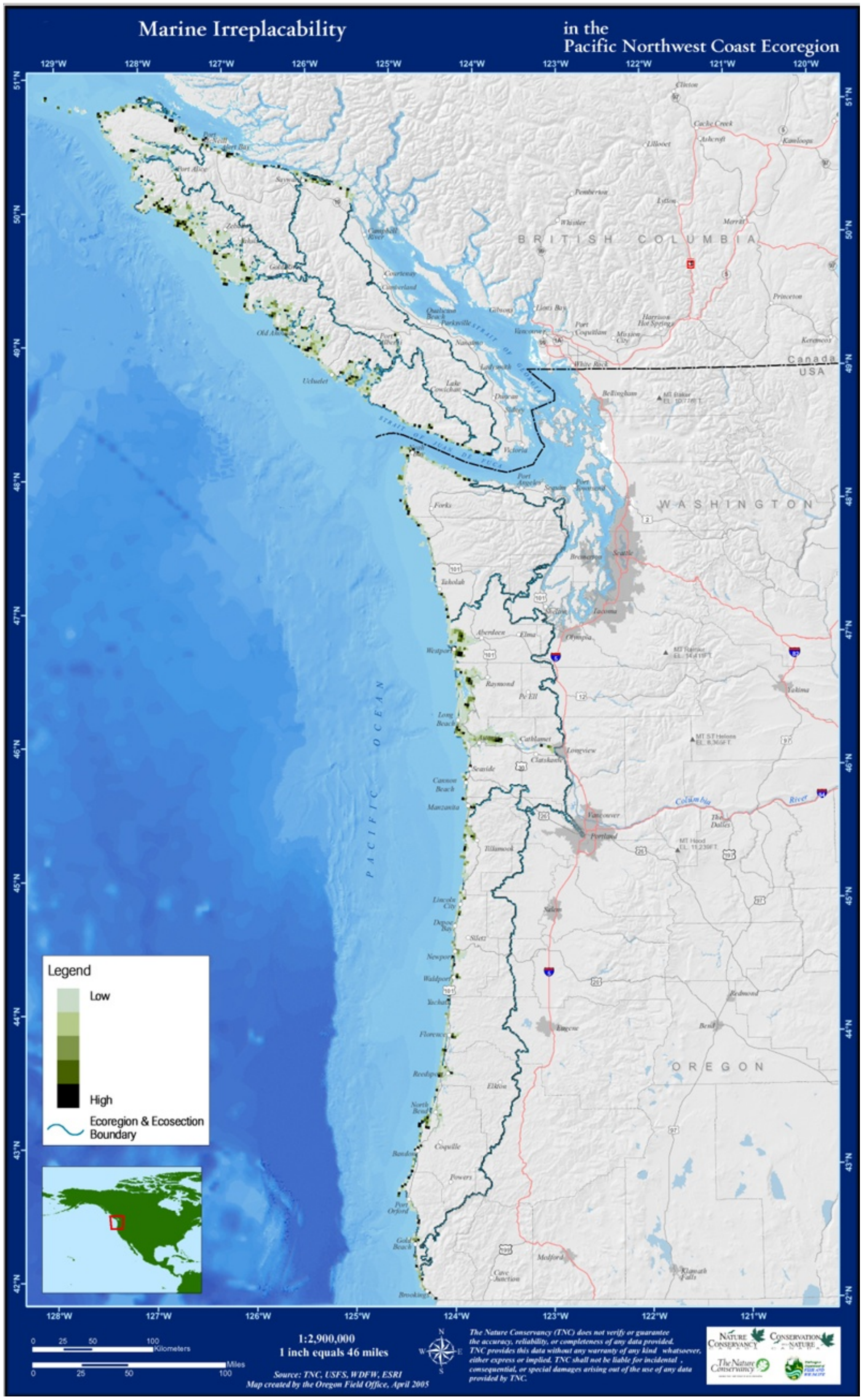
2.9.14 Nearshore marine analytical framework



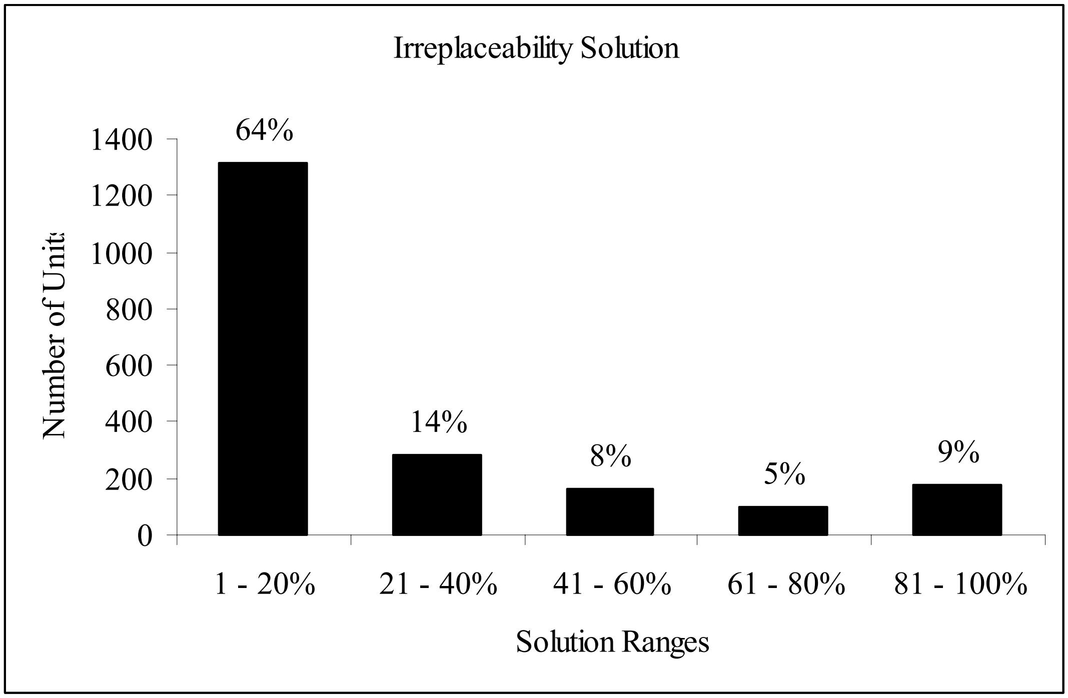
2.9.15 Nearshore and estuary planning units



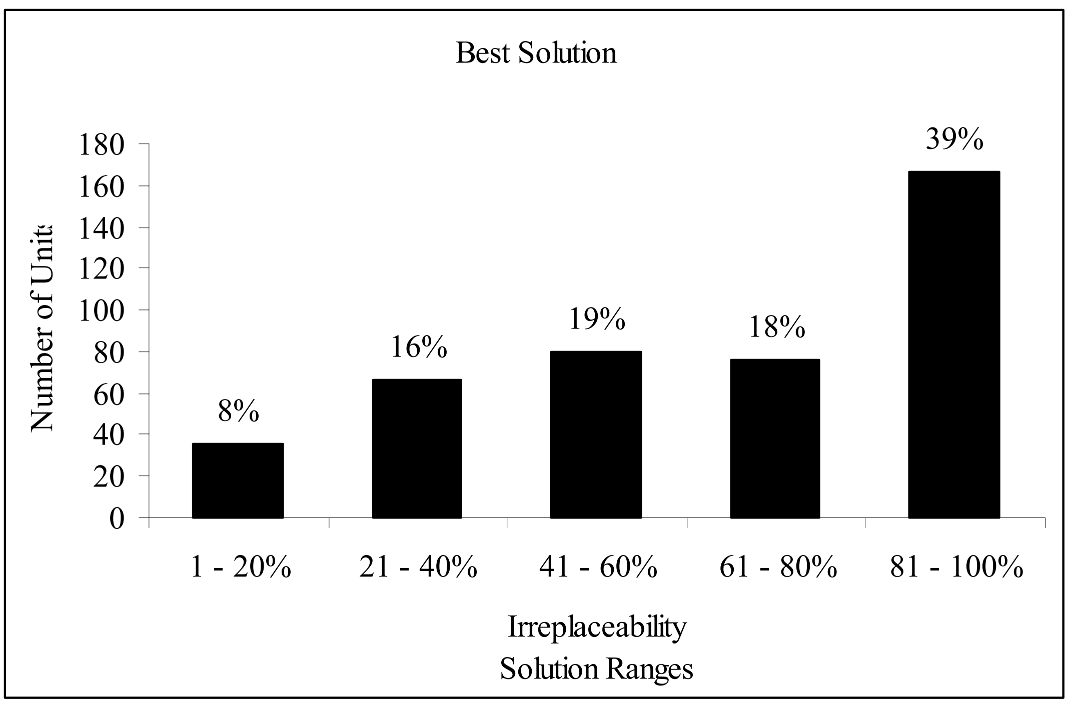
2.9.16 Irreplaceability analysis



2.9.17 Irreplaceability solution



2.9.18 Best solution



2.9.19 Area of conservation portfolio between the geopolitical units

Region	Region Area (ha)	Portfolio Area (ha)	% of Region
British Columbia	3,123,571	1,449,018	46%
Washington	2,194,156	989,189	46%
Oregon	2,852,533	1,126,554	39%
Total	8,170,260	3,564,761	44%

2.9.20 Length of shoreline in portfolio distributed between the geopolitical units

Region	Region Shoreline (m)	Portfolio Shoreline (m)	% of Region
British Columbia	5,934,242	1,988,084	34%
Washington	1,054,492	681,714	65%
Oregon	2,031,418	1,036,046	51%
Total	9,020,151	3,705,844	41%

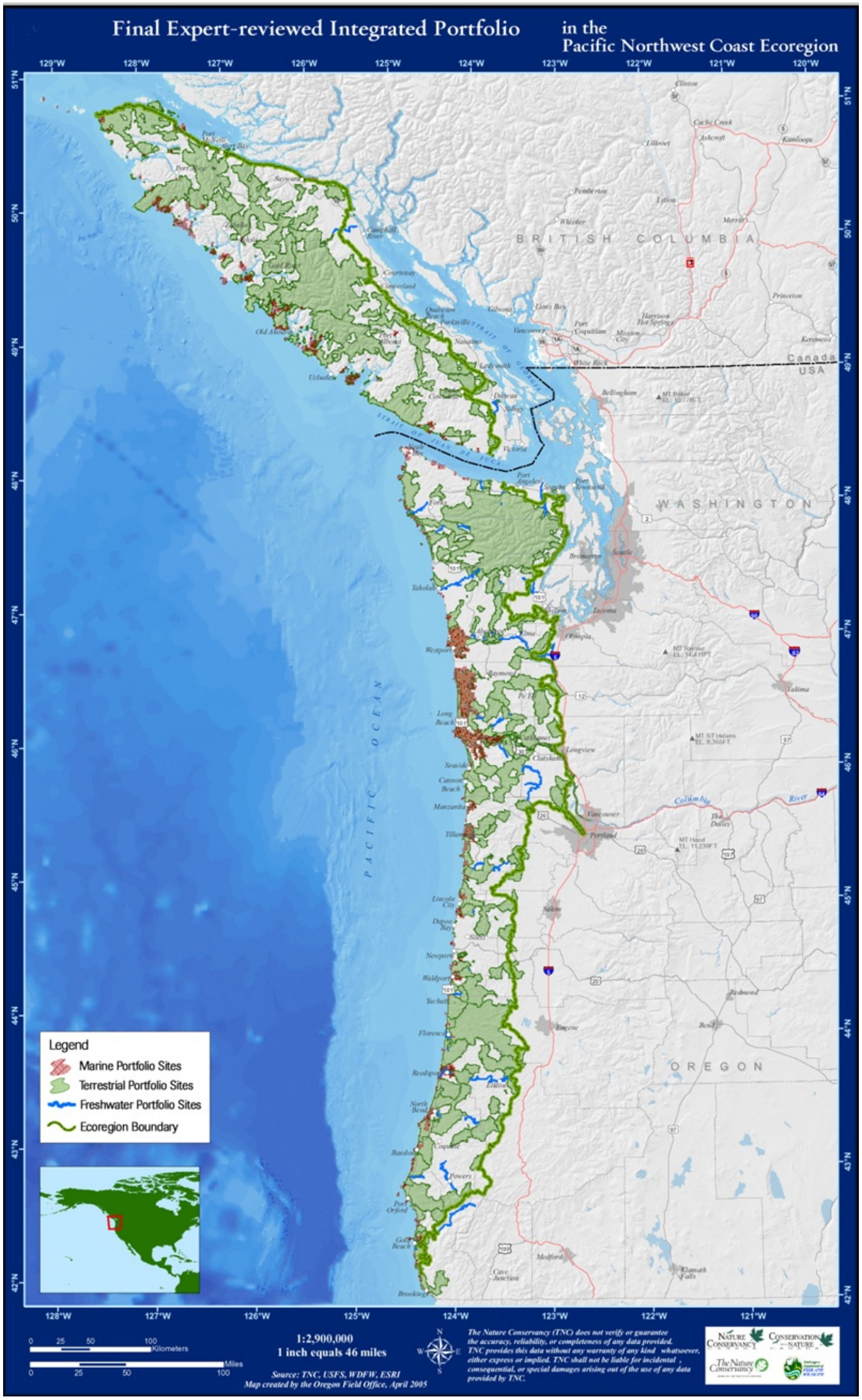
2.9.21 Portfolio area distributed between terrestrial ecological sections

Ecological Section	Section Area (ha)	Portfolio Area (ha)	% of Section
Nahwitti Lowlands	251,919	128,767	51%
North Isle Mtns	532,603	274,715	52%
Lee Isle Mtns	1,116,104	481,437	43%
Windward Isle Mtns	1,181,583	563,348	48%
Olympics	1,107,517	597,975	54%
Willapa Hills	1,562,769	581,723	37%
Coast Range	2,374,083	935,884	39%
Total	8,171,578	3,563,849	44%

2.9.22 Portfolio area distributed between terrestrial ecological sections

Marine Ecosection	Ecosection Shoreline (m)	Portfolio Shoreline (m)	% of Ecosection
Cape Arago North	1,619,959	823,874	51%
Cape Arago South	397,255	198,552	50%
Juan de Fuca Strait	274,759	142,632	52%
Johnstone_Strait	212,494	108,221	51%
Pt Grenville North	256,086	118,315	46%
Pt Grenville South	705,171	514,160	73%
QC Sound	65,701	38,871	59%
QC Strait	188,818	108,875	58%
VI Shelf	5,299,634	1,652,163	31%
Total	9,019,877	3,705,663	41%

2.9.23 Final integrated conservation portfolio



2.9.24 Conservation targets captured in the portfolio

Target Group	total # of targets analyzed	# of Targets with Goals	# meeting SITES Goals	% meeting SITES goals	# meeting conservation goals	% meeting conservation goals
Ecological Systems	25	25	24	96	24	96
Aquatic Systems	408	388	327	84	324	84
Fishes	66	61	51	84	51	84
Vascular Plants	60	59	49	83	5	8
Nonvascular Plants	11	11	8	73	0	0
Herptiles	13	12	11	92	11	92
Mammals	16	8	7	88	4	50
Birds	19	15	12	80	9	60
Insects	16	11	11	100	3	27
Mollusks	10	8	8	100	4	50
WA wetlands	19	19	19	100	19	100
OR wetlands	20	20	19	95	19	95
Estuary Habitat	24	24	23	96	23	96
Subtidal Habitat	4	4	4	100	4	100
Estuary Shoreline	57	57	57	100	57	100
Coastal Shoreline	61	61	50	82	50	82
Marine Invertebrates	1	1	1	100	1	100
Marine Fish	3	3	3	100	3	100
Marine Mammals	2	2	2	100	2	100
Seabirds	12	12	11	92	11	92
Marine Shorebird areas	3	3	3	100	3	100
Mineral Springs	1	1	1	100	1	100
Shorebird Concentration Area	1	1	1	100	1	100

3.0 Building a Benthic Habitat Model as Surrogates for Ecosystem-Scale Targets

This section describes the first steps in developing and comparing models for mapping offshore benthic habitats in the Northwest Coast Ecoregion. This, like all benthic models, is a work in progress. We utilized a topographic model and existing classifications that characterize depth and benthic substrate to model and generate offshore benthic conservation targets. Use of the benthic habitat model assumes that benthic habitat types can serve as a surrogate or coarse filter for the conservation of the majority of bottom-dwelling species in an ecoregion. The ideal data for mapping marine ecosystems is biological data on the distribution and abundance of species in the water and on the sea bottom. Unfortunately, these data are scarce offshore.

Lacking regionally comprehensive biological data along the Pacific Northwest Coast (PNWC), the Conservancy has focused on the use of geophysical data. We predict that many geophysical variables (e.g., temperature, depth and sediment type) can be correlated with the occurrence of different types of species. Geophysical information that is most useful includes sea surface temperature, bottom temperature, depth, bottom sediment type, phytoplankton density (chlorophyll a), currents and bathymetry (underwater topography). Our current model presented here uses bathymetry and marine geology to depict depth, geomorphology or bedforms, and substrate type.

It is our hope that the benthic model will be predictive of habitat targets. Output of the model, however, needs to be tested against higher resolution data (i.e., multibeam) and underwater surveys to determine the accuracy of identifying landforms on the seafloor. In addition, these data need to be correlated with biotic assemblages in determining community or habitat types. A recent study used local population density estimates of juvenile demersal finfish from trawl survey data as a meaningful indicator of habitat value (Cook and Auster 2005). We believe associating species data with modeled data on benthic habitats will ultimately give us a more accurate spatial assessment of species-habitat utilization. Lastly, it should be noted that this model cannot be used to predict surface or water column patterns in diversity. Other models are required in examining the pelagic environment.

3.1 Classification of the Benthic Environment

In order to generate a continuous surface depicting the seafloor we used a number of regional bathymetric data sets and examined interpolation techniques. Digital Elevation Models (DEMs) of the seafloor are distinct from terrestrial models in that the survey efforts required to produce a continuous surface of depth across a region are often inconsistent temporally, spatially and methodologically. Therefore careful examination of interpolation methods was conducted before an appropriate surface was used to model benthic habitats.

After generating a continuous surface depicting the seafloor, we examined several models that classify the benthic environment into distinct geomorphic types. The benthic model presented here has been used for marine ecoregional planning throughout the continental

U.S., including the Southern and Northern California ecoregions, the Floridian and Carolinian on the east coast, as well as in the Northwest Atlantic Coastal and Marine region. In addition to developing an initial methodology and data for depicting benthic habitats we have also used the bathymetric source data to determine areas of bottom complexity. Although using the same source data, output from a complexity model complements the identification of benthic habitats and therefore will be addressed separately. Both methodologies were conducted along the outer coasts of Oregon and Washington, part of the Pacific Northwest Coast ecoregion (Figure 3.4.1).

The results of the benthic habitat model described below produce offshore marine conservation targets. This approach to modeling coarse scale habitats provides promise in areas of the world where comprehensive thematic mapping of the seafloor has not occurred. The benthic model combines three parameters: geomorphology, depth and substrate. We initially examined six different geomorphic types to describe the seafloor (basins\canyons, lower slopes, middle slopes, upper slopes, flats, ridges) but later combined all the slope position types into one. We then combined the four geomorphic types (basins\canyons, slopes, flats, ridges) with four depth ranges:

Class	Definition
Inner shelf	0-40m
Mid shelf	40-200m
Mesobenthal	200-700m
Bathybenthal	700-5000m

These depth classes were primarily based on Greene et al. 1999 but were also informed by others (Allen and Smith 1988, Zacharias et al. 1998). The modeling produced 16 potential bedforms (combined geomorphology and depth) which represented our initial list of benthic habitat types. The last step incorporated lithology or substrate. For the purposes of developing the benthic habitat model we identified the most common descriptions of bottom induration types: “hard”, determined from rock and boulders classes; “soft”, determined from sand or mud bottoms; or “unclassified”. With this combination of geomorphology, depth, and substrate there were 48 potential benthic habitat types.

3.2 Benthic Habitats

We applied a landscape position model described in Fel and Zobel (1995), and later described in detail by Weiss (2001) for mapping seafloor geomorphology. Since landscape classifications are not based on morphology alone but also on the position of the land surface in relation to its surroundings, Fel (1994) developed a quantitative index of landscape position. Also called Topographic Position Index, or TPI, the basic algorithm compares the elevation of a given cell in a Digital Elevation Model (DEM) to the mean elevation of a specified neighborhood around that cell. Positive TPI values represent locations that are higher than the average of their surroundings, while negative TPI values represent locations that are lower than their surroundings. TPI values near zero are flat areas. This model was created to describe landforms in the terrestrial environment, but is easily adaptable to marine data.

Topographic position is an inherently scale-dependent phenomenon. Scale of the source data and the landscape context are two important factors to consider when deciding the search radius of a specified neighborhood, or groups of cells evaluated in a specific GIS procedure (see Zeiler 1999 for a good explanation of geospatial terminology).

a) Scale of the source data determines the level of detail that the model can depict. For instance, if the search radius is small then features within a small geography will be explicitly depicted given detailed source data; on the contrary, if the search radius is large then features may be missed or dissolved into larger categories. This scenario can also be true if the search radius is smaller than the source data can support. In other words, if the search radius is relatively small for coarse scale data then errors in interpolation may be mistaken for distinct features. To avoid these potential miscalculations it is important to evaluate the scales of the source data and examine different search radii to determine appropriate output models.

b) Seascape context determines the position of a distinct feature in relation to its surroundings. For example, a point in a basin may be coded as flat when the search radius is small; with a large search radius that same point may be considered at the bottom of a canyon if the surrounding area contains steep slopes that rise dramatically. Therefore, the nature of the broader land or seascape needs to be considered when setting the search radius in order to accurately represent variation in habitat.

As a general rule, the continuum of topographic position values sort out along a topographic gradient from depressions and canyon or valley bottoms, through to lower slopes, mid slopes, upper slopes, and up to ridge and hilltops. By determining thresholds for the continuous values they can be classified into distinct slope position categories (Figure 3.4.2).

Many physical and biological processes acting at a given location are highly correlated with the topographic position: a seamount, basin or canyon, ridge, flat plain, upper slope, etc. These processes (i.e., soil deposition, hydrologic balance and response, wind or wave exposure) are often important predictors of vegetation and other biota. Physical processes are difficult to model directly across large areas, but an index of topographic position can be used within a statistical predictive modeling framework as a surrogate variable to represent the spatial variation of these processes. For this exercise we modeled benthic geomorphic types using the same principles and tools developed in terrestrial models (Figure 3.4.3). In both environments a cell-based DEM is required, with cell values either representing elevation (positive) or depth (negative).

Recently marine practitioners have adopted this method for deriving landforms, calling this the Bathymetric Position Index, or BPI (Rinehart et al. 2004). Although the BPI model derives landforms on the seafloor, we have added depth classes (Figure 3.4.4) and substrate types (Figure 3.4.5) that further delineate distinct marine formations.

These modeling efforts were based on bathymetry data from the National Oceanic and Atmospheric Administration (NOAA), Washington Department of Fish & Wildlife (WDFW), and the Ministry of Sustainable Resource Management (MSRM) in British Columbia, Canada. The main issues to consider when assembling a mosaic of disparate data include scale of the source data and the search radius in depicting seafloor morphology mentioned above. Bathymetry data yields both the benthic geomorphology and depth of that formation. We combined the geomorphology and depth data with lithology on the seafloor. The Oregon and Washington continental shelf geologic data set compiled and mapped by Oregon State University (Goldfinger et al. 2001) and others (Greene et al. 1999), as updated for the Groundfish EFH-EIS process, incorporates available information on seafloor substrate types for the region. In addition, geologic data was available for British Columbia (MSRM 2001). We used a simplified classification of marine substrate types (hard, soft, unclassified) in order to match data across the region.

The resultant grid after combining geomorphology and depth with substrate types tracked all potential combinations of inputs resulting in 48 (4 landforms x 4 depth classes x 3 substrate types) unique benthic habitat types for the Pacific Northwest Coast ecoregion (Figure 3.4.6). A final check was conducted to determine whether all 48 modeled benthic habitat types were present in the ecoregion; a few types were present but at <100 total hectares (inner shelf canyon unclassified (1.2 hectares), inner shelf slop unclassified (53.6 hectares), and mid shelf canyon unclassified (82.2 hectares)). The largest category was bathybenthal flats unclassified (3,725,682.2 hectares); the total area cover was 14,716,641.8 hectares from mean high water to approximately 2,500 meters depth.

It should be noted that these categories were also used in the Northern California Coast ecoregion and therefore could be combined to illustrate Pacific west coast-wide coverage (TNC 2005).

3.3 Roughness of the Seafloor

The Global Marine Initiative of The Nature Conservancy has completed an initial analysis of seafloor roughness, or rugosity, along the outer coasts of Oregon, Washington, and Vancouver Island. This work is a continuation of ongoing terrestrial, freshwater, and nearshore ecoregional planning efforts in the Pacific Northwest Coast ecoregion. Funded by an 18-month cooperative agreement between The Nature Conservancy's Global Marine Initiative and the National Oceanic and Atmospheric Administration's Coastal Service Center - GIS Integration and Development Program - the goal of the project is to improve methods for identifying priority sites for marine conservation and management action. This is being done through spatially explicit regional planning that unambiguously accounts for the vital connections between land, rivers, and sea.

Many taxa are strongly correlated with areas of habitat structure and complexity of the sea floor (Beck 2000, Yoklavich et al. 2000, 2002, Hixon et al. 1991, Field et al. 2002, Starr 1998, and Williams and Ralston 2002). These site level studies have verified the presence of more abundant and more diverse assemblage of species in areas where rocky reef or other hard bottom features occur. Still, there is some confusion among conservation planners as to the correct terminology associated with complexity and rugosity. The term complexity refers to the specific habitat needed for individual species or species groups. Take rockfish for instance. Rockfishes tend to inhabit areas with various amounts of hard complex strata (i.e., rock ledges, caves, crevices, boulders, cobble fields) and other vertical structures (like kelp forests and various macroinvertebrates) during at least part of their lives (Love et al. 2002). These habitat variations refer to the horizontal and vertical rocky structures, or complexity, on the seafloor. Complexity is related to but not the same as relief, which refers to the maximum change in depth. While some benthic features can be modeled and are therefore considered complex (i.e., cobble fields) others cannot (i.e., caves, crevices).

However, use of the term complexity is wide spread and is often confused with rugosity. For many modelers and planners complexity is measured by how often the slope of the sea bottom changes in a given area (see Ardron and Wallace 2005). In the modeling world this refers to the density of the slope of slope (second derivative) of the depth. Because this habitat modeling is usually done with depth sounding or multibeam data we would argue that this cannot depict vertical rocky structures and other complex features. We have observed that output of these models refers to the roughness of the seafloor, or changes in depth over an area, and not as topographic complexity. The term rugosity, therefore, may be a more applicable term to describe this kind of habitat modeling. The ratio of surface area to planar area is a measure of rugosity or roughness (see Iampietro and Kvitek, Jenness 2002). Rugosity literally means "that state of being thrown into folds or wrinkles." It is also described as a roughness factor of a surface, where the real surface of an area is divided by the geometric surface of that area. We therefore use the term rugosity to describe our habitat modeling efforts.

Recently larger scale efforts have been conducted to model the seafloor in order to depict areas of benthic rugosity (TNC 2005, NCCOS 2003, Ardron 2002, MSRM 2001). These modeling efforts often occur where there is insufficient data to explicitly map sea bottom features. This work by different government and non-government agencies is being done for a variety of reasons, but they all share an objective to highlight distinct features of benthic habitats. We examined four studies that calculated the roughness of the sea floor to determine which method or methods would best apply to marine conservation planning efforts in the Pacific Northwest Coast ecoregion. The four studies included:

1. The Duke University Geospatial Analysis Program (DUGAP) that has been partnering with The Nature Conservancy in its efforts to complete ecoregional assessments in South Florida and South Carolina, Floridian (Eschelbach et al. 2005) and Carolinan (DeBlieu et al. 2005) ecoregions respectively.

2. The National Oceanic and Atmospheric Administration's (NOAA) National Ocean Service (NOS) and the National Center for Coastal Ocean Sciences (NCCOS) have conducted a biogeographic assessment of Northern and Central California (NCCOS 2003)

3. Living Oceans Society (LOS), a non-profit organization in British Columbia, has a couple of rugosity studies off the north and central coasts as well as Vancouver Island (Ardron 2002, Rumsey et al. 2003)

4. The Ministry of Sustainable Resource Management (MSRM) in British Columbia contracted a study to model rugosity as part of a larger project to develop benthic and pelagic ecounits (MSRM 2001)

We initially ran each method separately and compared results. The NCCOS approach represents a simplified method of mapping the seafloor and correlating the results with data on fish assemblages (NCCOS 2003). This method contained the fewest process steps, focusing primarily on calculating the standard deviation on the original bathymetry data. This calculates the standard deviation of every cell in the original bathymetry with a circular radius of 33 cells. Our cells were 30 meters, so this approximates a radius of 1 kilometer. This was adopted directly from their biogeographic assessment method to calculate seafloor roughness. After calculating standard deviation we had to establish breaks in the data in order to isolate specific areas of complexity. We used one standard deviation from mean to break the data into distinct classes. Classes were established by identifying standard deviations at or less than 1 (relatively flat or smooth undulations on the seafloor), values of 1 - 2 ("low rugosity"), and values of 2 - 3 ("high rugosity"). The last step was to convert the classified standard deviation grid into polygons. This was done in order to establish minimum size thresholds of the different classes and compare these areas with the more elaborate approaches to calculating roughness. This simplified approach is now being applied to NOAA's biogeographic assessments throughout the United States.

In British Columbia, Canada, both government agencies and non-profits have been conducting rugosity studies at regional scales. The Ministry of Sustainable Resource Management (MSRM) has commissioned studies to establish ecounits, or distinct areas in benthic and pelagic environments based on seafloor and oceanographic characteristics, respectively. These were established in order to design policy and manage resources within these units. Non-profit research and analysis of seafloor roughness has been conducted in efforts to identify areas of high benthic diversity and implement conservation strategies in those areas. Both approaches consider the calculation of "slope of slope" the core step in determining “topographic complexity” (see Ardron and Wallace 2005). These methods will not be discussed here, but steps adopted from these studies in our modified approach will be detailed below. For in depth treatments of these analyses see MSRM 2001 and Ardron 2002.

The Duke University Geospatial Analysis Program (DUGAP) has been researching methods for calculating rugosity in efforts to help The Nature Conservancy complete ecoregional assessments along the east coast of the United States (Eschelbach et al. 2005, DeBlieu et al. 2005). This approach offers an alternative to calculating slope of slope, where the variety of slope values in a determined area serves as the initial step. After examination we determined that this approach, with modifications, was the preferred method for our analysis of the benthic environment in the Pacific Northwest Coast marine ecoregion.

Preferred method

The DUGAP method was modified but utilized specific steps from Living Oceans Society approach for calculating rugosity. A combination of these methods determined our preferred approach. The first step was to calculate the variety of cells in a specified neighborhood, or groups of cells evaluated in a specific GIS procedure (see Zeiler 1999 for a good explanation of geospatial terminology). This returns a range of values, where the upper 50% of the most variety is classified as rough. It should be noted that there are many ways to classify these data, both in terms of classification scheme used (i.e., natural breaks, quantile, equal interval) and the number of breaks in the data. It is best to examine different schemes and breaks to identify the upper 50% of high variety. The second step was to subtract the original bathymetry values from derived bathymetry data where the mean was calculated in a specified neighborhood. This identified areas where there was a significant difference between original and mean values on a cell by cell basis. Similar to identifying the upper range of variety, the upper range where significant differences were found were then classified to determine roughness. The third step using this method combined the variety and value difference grids. Cells could then be selected if a) they had low variety but a large difference from mean elevation, b) they had low difference from mean elevation but high variety, or c) they had high variety and high difference from mean elevation.

This output was then converted to points in order to calculate the density of values after combining variety and difference from mean grid data. The search radius setting is the most important parameter to examine, as results will vary depending on the set neighborhood. Since this is scale dependent it is best to determine the radius after evaluating the scale(s) of your original bathymetry grid. If you have combined widely ranging scales for depth values across the study area, you may want to calculate point density in sections. Generally, the larger the scale (coarser the data) the higher the radius. Too small a radius might extract details not found or known in the seascape, and too large a radius might over-generalize the features that the original data support. We used 1000 meters, or a kilometer, to calculate point density. This step of converting grid cells to points and calculating density, and the subsequent step, was adopted from Living Oceans Society (Ardron 2002).

Similar to other process steps, the goal was to isolate specific areas of rugosity that the data support (Figure 3.4.7). For the Pacific Northwest Coast we used one standard deviation from the mean to break the data into distinct classes. Standard deviations at or less than 1 were not considered rough; values of 1 - 2 were considered "low rugosity," and values of 2 - 3 were considered "high rugosity". The last step was to convert the density grids to polygons. The resultant polygons had a range of sizes, and a threshold was established according to minimum area. If a high rugosity polygon was less than 100 hectares we either dissolved it into a surrounding low rugosity polygon, or it was deleted. If a low rugosity polygon was less than 100 hectares it was deleted. This helped ensure that site selection processes identify concentrations of rugosity areas.

Next steps
Having recently completed our examination of the four studies and the analysis in the Pacific Northwest Coast, we are now conducting a review of our proposed methods. Having constructed a modified version of the DUGAP method as our preferred approach, we compared these results to ones from the NCCOS method. We did not use any specific methodological steps from the MSRM study, although some of their processes were very similar to the other approaches.

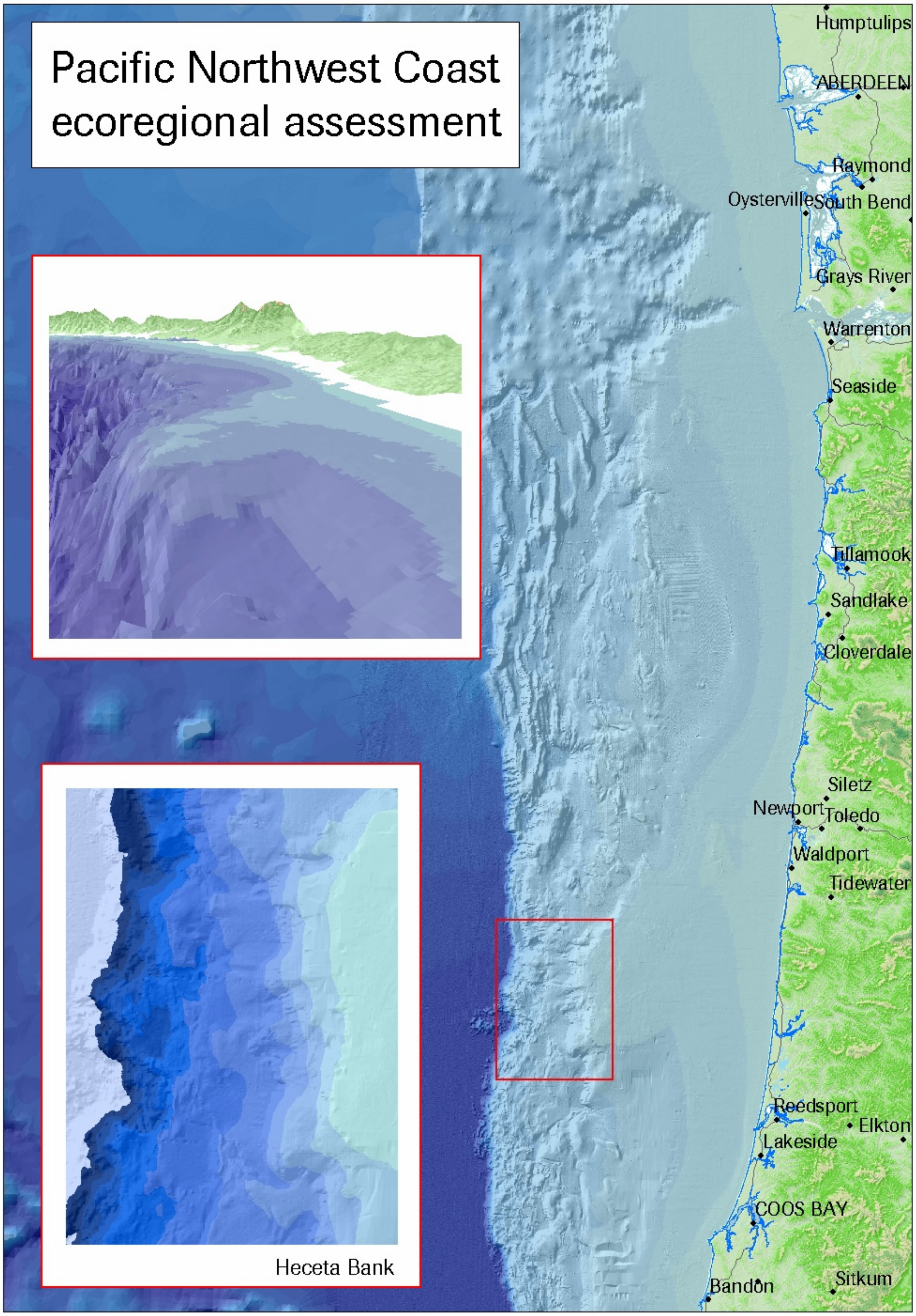
While there was significant overlap in areas identified as "low and high rugosity" between our preferred approach and the NCCOS method, results varied too widely across the region when comparing these to the Living Oceans Society method. The similarities between our preferred approach and NCCOS results allowed us to conduct quantitatively summaries in terms of identified places and area selected. We found that our approach produced more low rugosity areas, but fewer high rugosity ones. In addition, this approach identified more total area of rugosity and more explicitly isolated what we defined as high rugosity areas. Further, our approach yielded more spatially explicit areas where the sties were more defined compared to the general areas established with the NCCOS method. We also used the Living Oceans Society and MSRM results from their studies off of the West Coast of Vancouver Island to compare with our results.

We are now in the process of further examination of our results and testing other approaches (Jenness 2002). The next steps for this work will include a detailed comparative analysis between our preferred approach and NCCOS results, ancillary data to compare or "ground truth" the results of our analysis, and comparisons to the Jenness method which is the method embedded in the Bathymetric Position Index tool developed by Oregon State University and NOAA's Coastal Services Center (Rinehart et al. 2004).

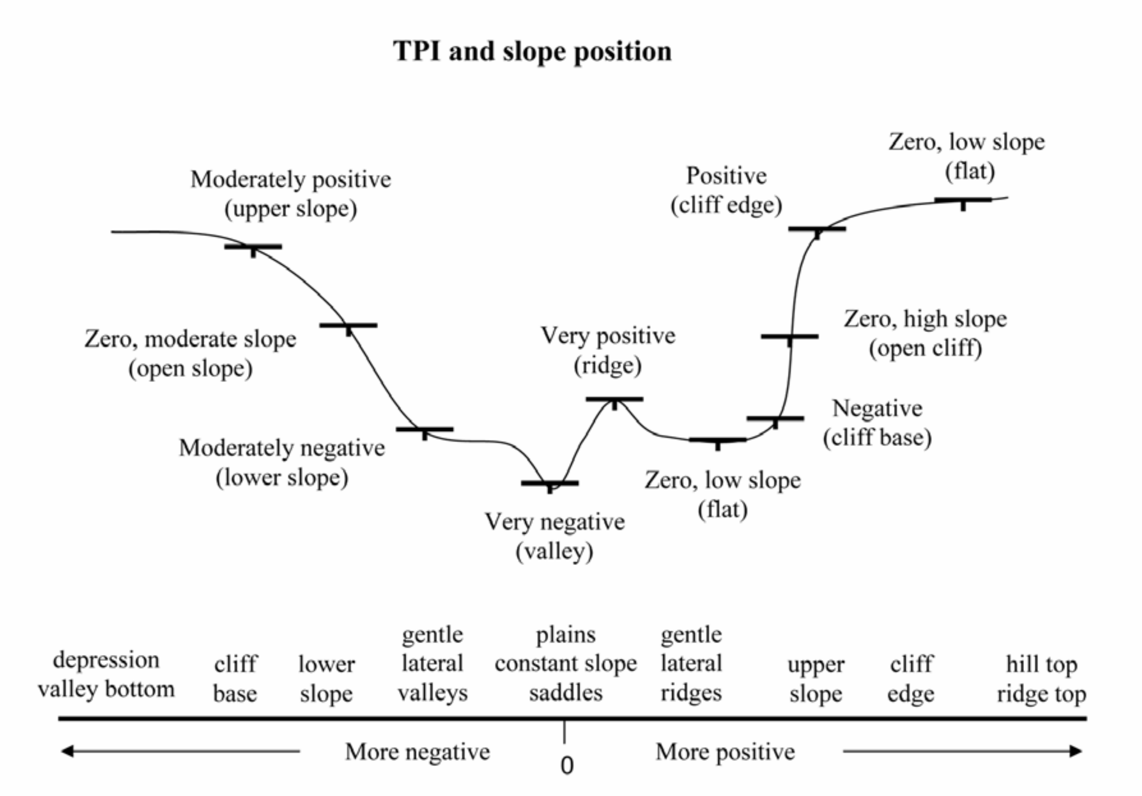
The analyses presented here are provided as a proxy for quantifying structure at a region scale (i.e., mesoscale rugosity), and should be interpreted with care as they represent an estimate of benthic rugosity at a scale of one kilometer. Similar analyses can be performed for an infinite set of ranges, each resulting in similar patterns with dimensions proportionate to the prescribed search radius of the cell neighborhood.

3.4 Figures

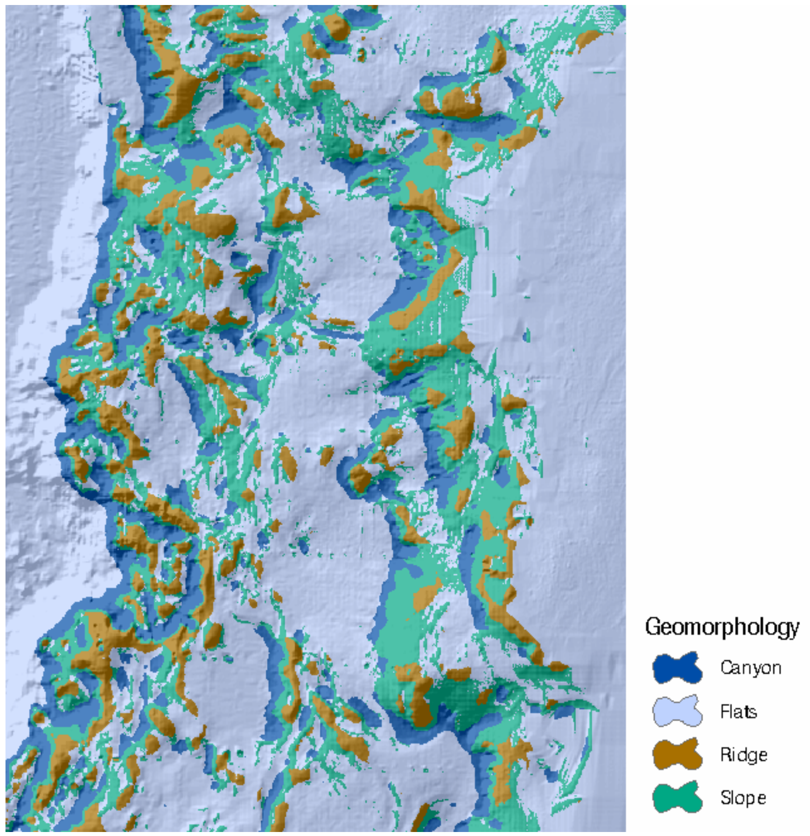
3.4.1 Bathymetry off of Oregon, Pacific Northwest Coast ecoregion



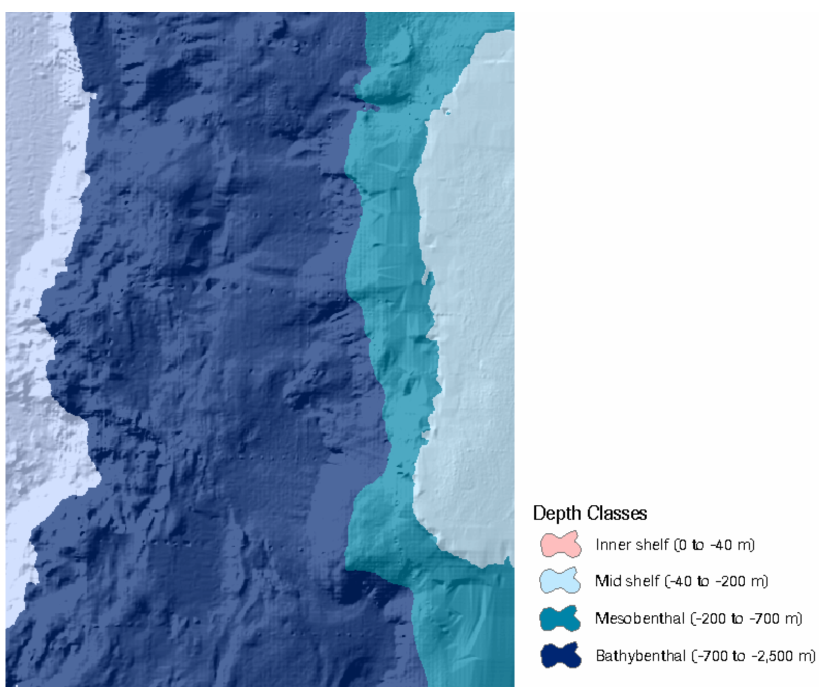
3.4.2 Topographic Position Index (TPI) models specific land or benthic features along a gradient of continuous values



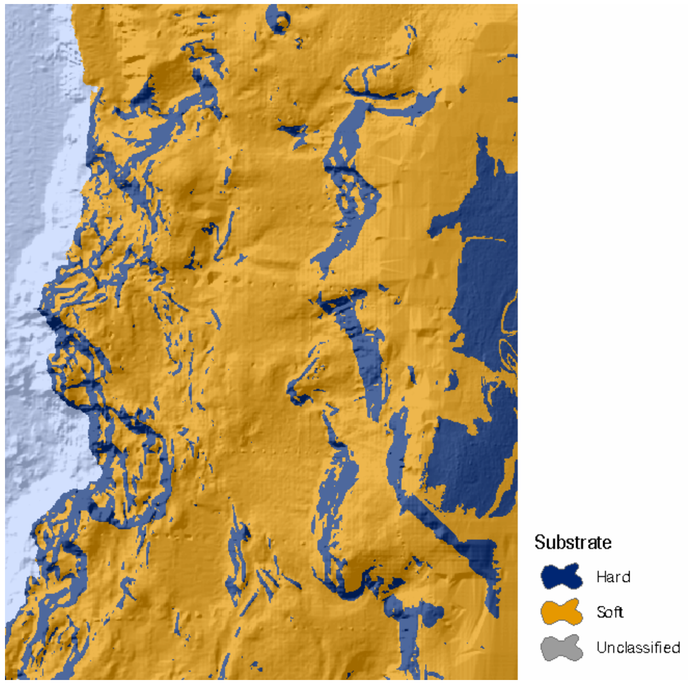
3.4.3 Geomorphic types on the seafloor for Heceta Bank off the southern Oregon coast



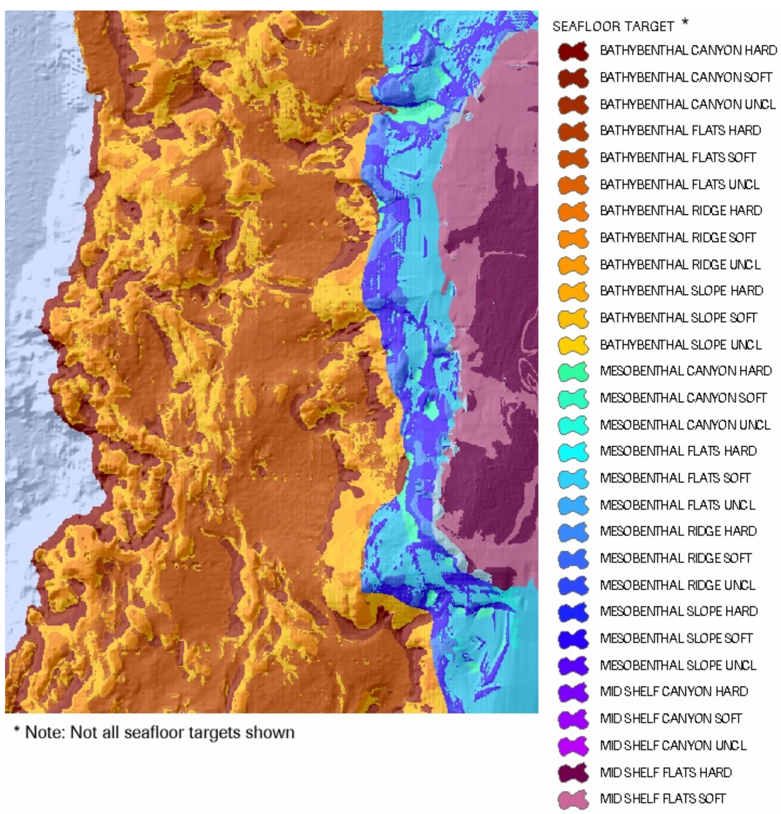
3.4.4 Depth classes for Heceta Bank off the southern Oregon coast



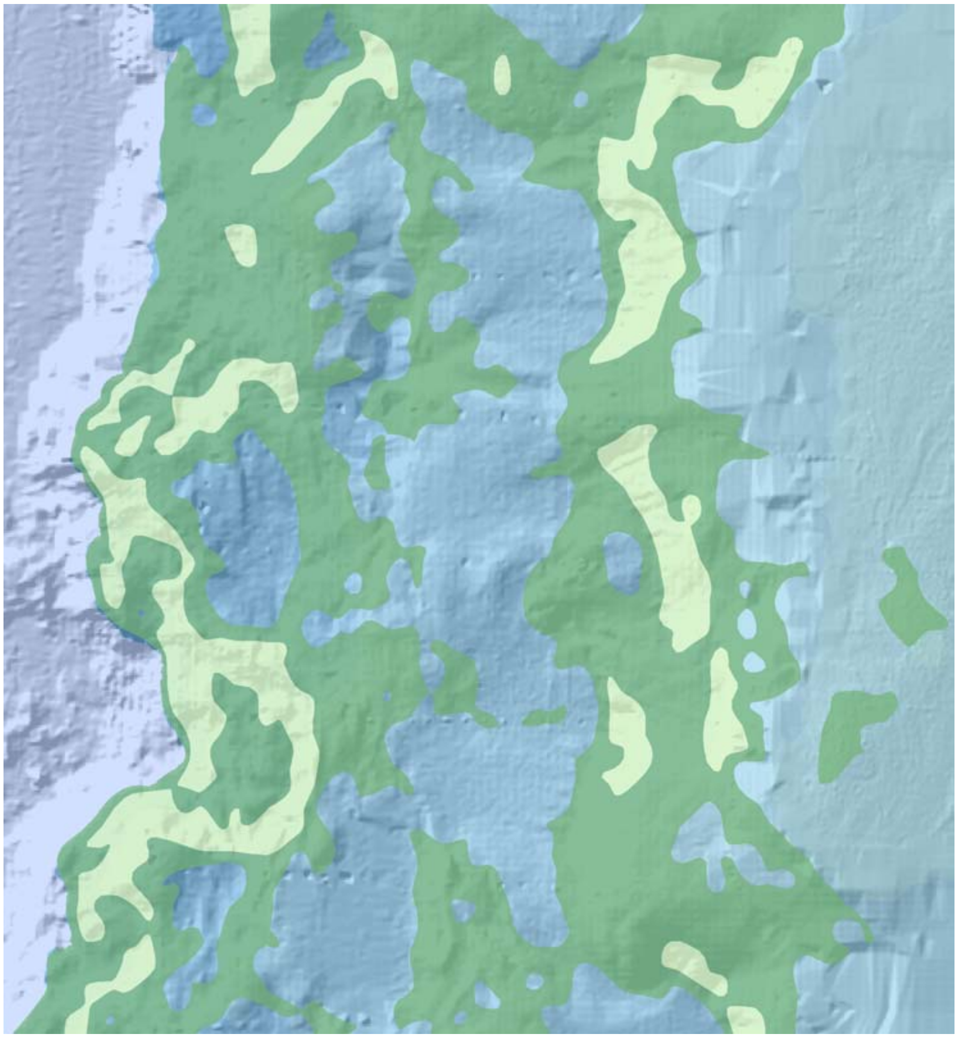
3.4.5 Substrate types for Heceta Bank



3.4.6 Final benthic habitat types for Heceta Bank



3.4.7 Identified areas of low (dark green) and high rugosity (light green) on Heceta Bank



4.0 Incorporating a spatially-explicit, land\sea threats analysis in the Pacific Northwest Coast

Abstract

There is a lack of quantitative spatial analyses addressing the problem of land and marine-base threats for integrated terrestrial and marine planning. These threats significantly alter land and seascape conditions and are therefore important to consider in conservation planning efforts. We have mapped and combined specific threats in the Pacific Northwest Coast ecoregion into a single index where those threats occur and utilized a decision support tool for evaluating the effects of different indices on priority sites selected for conservation. We compared three different approaches to weighting individual threat factors in the index and quantified their influence on site selection, including one without the inclusion of any factors. Threats included shoreline armoring, road density, and land cover. We have shown that there are numerous threats that affect priority setting in terrestrial and marine environments. We examined spatial variability and efficiency output, from the decision support tool MARXAN, and conclude that while variations in the construction of weighted indices did not significantly change the results, the inclusion of specific threats significantly altered the solutions. Including threats influences site selection analyses by forcing the algorithm to choose fewer sites which are more consistently selected. There is less variability in the result, where the algorithm avoids areas with a less desirable condition or a higher degree of threat. Selection results using single or multiple threats can be illustrated quantitatively, allowing comparisons between indices. We believe a thorough investigation of what threats to include in an index is important when considering conservation priorities, and that more work needs to be done to accurately account for threats that occur across the land/sea interface in order to advance the approach of integrated priority setting.

4.1 Introduction

The coast and nearshore marine environments experience both land and marine-based threats caused by human activity. Salzer and Salafsky (2005) define a threat as "any human activity or process that has caused, is causing or may cause the destruction, degradation and/or impairment of biodiversity and natural processes". A threat may represent a specific condition, trend, or seasonal variation in the environment that impacts a species, habitat, or ecosystem's ability to persist over time at a particular location. Threats here refer to current impacts in the environment that can be mapped, and their potential for ongoing or increasing effects.

Recent reports addressing these threats indicate the vulnerability of marine ecosystems, habitats and species to current and ongoing human influence (Pew Oceans Commission 2003, U.S. Commission on Ocean Policy 2004). Impacts to coastal ecosystems arise along the shoreline, in adjacent watersheds and in the marine environment by coastal currents and upwelling. Existing impacts and ongoing threats are deteriorating conditions along many U.S. shorelines and need to be addressed at site, landscape, and regional scales. The Pew Oceans Commission (2003) recently conducted the first national review of ocean policies in more than 30 years. They found a shared sense of urgency and commitment to reverse the decline in the health of the oceans, identified major threats to the ocean, and made recommendations to change ocean policy in order to more actively address these threats. Similarly, the U.S. Commission on Ocean Policy (2004) released a report identifying key threats to oceans and coasts. These reports, however, did not conduct any quantitative analyses of threats that illustrate their effects on the coastal environment nor recommend how others might address threats in specific geographies. There have been, however, qualitative analyses of threats and comparisons of individual threats (Jackson et. al 2001, MCBI 2003, Halpern (in press)). Although these studies have helped track the geographic extent and community-wide impacts of individual threats on species, habitats, and ecosystems, presently there are few quantitative representations of what constitutes a marine threat and where those threats are located.

As the majority of coastal and marine studies have focused on qualitative reviews identifying ocean threats, others have used geographic information systems (GIS) to determine environmental conditions across landscape or seascape level geographies (Reefs at Risk 1998, Cook and Auster 2005). Spatially-explicit analyses like these have made important steps in our understanding of the extent and level of impact individual threats have on the marine environment. One way to address coastal threats further is to incorporate this information into conservation planning exercises. The Nature Conservancy and its partners have adapted an ecoregional planning approach that combines biological and ecosystem-level information utilizing optimized site selection algorithms (Possingham 2000, Andelman 2000) to identify regional conservation priorities (Groves 2003). A core component of this approach includes a consideration of threats.

The conservation science literature has often referred to suitability when discussing environmental conditions (see Hopkins 1977 and Collins et al. 2001 for reviews), and there are many different methods for constructing a suitability index (Banai-Kashini 1989, Carver 1991, Miller et al. 1998, Stoms et al. 2002). Places where the most direct impacts occur represent areas that are considered the least suitable for conservation. For example, places where high intensity coastal development and road density occur adjacent to a highly structured shoreline represents the least suitable places to put conservation efforts. However, it is important to note that this does not preclude restoration and does not in any way imply that restoration efforts won't resuscitate the ecosystem in that area.

Generally, a suitability index also includes management designations that help offset these degrading conditions depending on the protective status of the area. These factors, both negative impacts and protected areas, have a significant influence on optimized site selection algorithms. For the purposes of this work we only combined threat factors into a "threat index" since we did not consider marine managed areas as additional factors. This was done to be able to clearly examine the influence of threats in site selection without the influence of political designations. The purpose of the threat index is to balance the desired selection of targeted species, habitats, and ecosystems with known impacts from threats in identifying conservation priority areas.

Determining and Quantifying Threats

One of the objectives of conservation planning is to identify places that provide adequate species diversity and ecosystem health to the larger landscape. This often has to be balanced within the human landscape. As people increasingly inhabit and impact the environment, conservation planners must account for direct and indirect human-related impacts while identifying those places that contain biodiversity and provide ecosystem services. We have considered three aspects of threats when deciding what factors to include in a regional analysis: a) the geographic extent, b) level of community-wide impact, and c) the urgency for abating threats.

First, identifying the scale of a threat is an essential component in evaluating its impact across land and seascapes. Threats in coastal areas (e.g., increasing human populations within 50 miles of the coast, coastal development) are direct impacts to coastal ecosystems that play out at relatively local scales. More distant human activities on land and in freshwaters have significant, although often overlooked, effects on coastal and marine ecosystems (Beck 2003). Threats originating in watersheds that link the land to the sea can traverse very large distances (e.g., nonpoint source pollution). Second, threats that have a community-wide impact are ones that likely degrade multiple species or habitats in many places. Within the ecoregional planning framework threats are usually evaluated per conservation priority area and not for an individual species or habitat (Groves 2003). This is due to the fact that we are just beginning to relate the scale and extent of individual impacts and threats to specific ecosystems (see Halpern et. al forthcoming). For example in the offshore, overfishing involves the direct take of targeted individuals, other species through bycatch, and habitat communities through fishing practices such as bottom trawling (Pauly 1998). This example clearly illustrates the direct relationship between threats and species or communities. Third, urgency for abating individual threats is determined by the likelihood of a threat negatively impacting an area within a set time frame. This requires qualitative determinations and is therefore subject to disagreement among stakeholder groups. Given that the shoreline environment in particular is subject to both land and marine-based threats, it has been difficult for expert groups to rank the severity of multiple impacts (see Floberg et al. 2004). Nevertheless, we have assessed trends in individual threats to determine whether they are ongoing or increasing.

We have used the Pew and U.S. Commission reports, in addition to other sources (Ervin and Parrish 2004, Salafsky et al. 2002, Salafsky et al. 2003, Salzer and Salafsky 2005, Sutter and Szell 2004, WDNR 2000, EPA NCA 1999 - 2000) in our process to determine threats in the analysis. We focused primarily on land and marine-based threats that have direct effects on the coastal environment. The choice of these effects depend on the cited threats in a given region and the availability of spatial data to map or model them (Stoms 2000). Spatial criteria include whether the factor is or can be made explicit enough to decipher patterns or possible trends of a current condition compared to a previous or presumed natural state. In general threat factors were considered here if they a) occur over the entire coastal environment, b) affect a greater proportion of species, habitats, and ecosystems, and c) are ongoing or increasing. We evaluated a variety of threat factors relevant along the outer coasts of Oregon and Washington, and conducted literature and data searches on their geographic extent and level of impact to biological features. We were limited by available spatial data at this scale, but were nonetheless able to identify shoreline armoring, road density, and specific categories of land cover as major threats.

What we outline here is an integrated land-sea approach to mapping threats in an analysis within the Pacific Northwest Coast ecoregion (Figure 4.5.1). We will examine several approaches to constructing a threat index and quantitatively test its influence on setting conservation priorities. These include 1) an index with equal values for every planning unit which does not include threats, 2) a relative value index where individual threat metrics are ranked and weighted, and 3) an absolute value index that utilizes the actual metrics of the threat (e.g., length of shoreline armoring as value). In the second and third approaches it is assumed that the threat factors are equal in impact or importance; threat factors were not weighted relative to one another.

4.2 Methods

Marine planning at the eco-regional scale provides a larger context for selecting high priority conservation areas in estuarine, nearshore, and offshore environments. Ecoregions, not political boundaries, provide a framework for capturing ecological and genetic variation in biodiversity across a full range of environmental gradients. Defining and tracking the influence of threats also requires a delineation of the land-sea boundary within the Pacific Northwest Coast. We utilized ecological drainage units defined by major basin boundaries with similar biotic patterns that conform to physiography, climate, and freshwater ecosystem connectivity (Higgins 2005). They are derived from subsections of World Wildlife Fund freshwater ecoregions (Abell et al. 2000). These units were used to locate the watershed area that contributes freshwater to the coastal marine environment. In a few cases (e.g., Columbia River) the ecological drainage unit extended beyond the boundary of the ecoregion. Here we measured the extent of freshwater influence from the entire drainage unit, but conducted site selection only within the ecoregional boundary (Figure 4.5.1).

For the marine side of the analysis we used the 40-meter isobath as the seaward boundary. This was the same extent used in the ecoregional assessment, roughly approximating the zone of light penetration along the outer coast. Although the zone of freshwater influence extends beyond this depth for the major river systems we limited the analysis to the coastal and nearshore waters largely due to our focus on intertidal and shallow subtidal conservation targets. We were also interested specifically in how land-based threats affect adjacent marine waters before getting dispersed by major current systems (i.e., California Current).

This delineation yielded 4,956,000 hectares within the study area, or 3,943,000 hectares of land and 1,013,000 hectares of marine waters. Broken out into 500-hectare assessment units there were 7,886 land units or 80% coverage of terrestrial targets and threat factors, and 2,026 marine units or 20% of the analysis containing marine targets and threats information.

This planning region then underwent a regional process of identifying and analyzing four key components: conservation targets, conservation goals, ecological integrity and the selection of high priority conservation areas. These are briefly outlined below. For a more in depth treatment, see Beck et al. (2003).

Conservation Targets

The first step is to select conservation targets. These are ecosystems, habitats, and species that represent a diversity of the biotic assemblages in a region. In marine environments the most effective planning approach is to focus on marine ecosystems and the ecological processes that sustain them (Beck et al. 2003). This presumes that the conservation of representative ecosystems will also conserve the diversity of species found in these ecosystems. Examples include rock platforms that support tide pools, kelp forests, and seagrass meadows. A robust classification scheme to identify the different types of ecosystems is critical for selecting conservation targets. Where possible classification schemes should be based on biological data, but in the marine environment surrogate data is usually required, such as landform, slope, and wave energy.

Marine species targets that are least likely to be represented by ecosystem level information are endangered, imperiled, or species considered keystone (i.e., Power 1996). Many of these species require individual attention because management of their habitats alone is necessary but insufficient for their conservation needs. In addition, life stage information including the spawning aggregations of reef fish or breeding congregations of seals and sea lions on haulout sites are important to specifically target.

For this analysis we examined 520 conservation targets, where 315 were considered estuarine and nearshore marine and 205 were terrestrial. These targets have been stratified into subregions to be sure all the diversity is represented. These subregions are delineated based on physical parameters including bathymetry, currents, salinity, and sea surface temperature. With stratification we are primarily trying to 1) represent unknown biodiversity (e.g., possible genetic variation in species or community level variation in ecosystems) and 2) spread sites out to avoid local catastrophes (spread risk; ensure replication). Previous to stratifying the marine region we identified 191 targets (58 shoreline types, 18 estuarine types, 26 intertidal vegetation habitats, 25 marine fish, 39 seabirds and shorebirds, 12 marine mammals, and 13 marine invertebrates).

Conservation Goals

A conservation goal identifies the amount of the target that should be represented in conservation areas across the planning region. The objective is to assess how much representation is required to maintain its persistence over time. This should ideally be based on historical estimates of the abundance and distribution of the targets. Unfortunately, goals often have to be based on current distributions (i.e., Beck and Odaya 2001). This being the case, different approaches have been adopted to test the representation question in marine environments (i.e., Leslie et al. 2003). One such approach is to conduct sensitivity analyses. This involves systematically varying the conservation goals to determine how they affect the overall size of the area selected.

Ecological Integrity

As we gather data on the distribution of the targets and note their locations, we attempt to ensure that we only include populations of species and examples of ecosystems that are likely to persist into the future (Beck et al. 2003). However, formal analyses of viability are rare for marine species and similar analyses of integrity are virtually non-existent for ecosystems. While we may not have these sources of information there are often factors that can be used to screen or filter out areas that are not likely to have the best or most viable examples of species and ecosystems. These are often called "cost" factors (Ball and Possingham 2000), but for this paper they are referred to as threats. These threats are assembled into a threat index, combining each spatial element into an aggregate planning unit and associated value. Threats included in this analysis are shoreline armoring (seawalls, jetties), land use designation (urban coastal development, agriculture), and road density (Figure 4.5.2).

Shoreline armoring

These data came from two sources, the Washington Department of Natural Resources (WDNR), and the Oregon Department of Fish & Wildlife (ODFW). The WDNR data was derived from helicopter surveys over the shoreline where an inventory was done to attribute the percentage of armoring to a shoreline segment (an individual segment defined by homogenous beach substrate). This data set is called ShoreZone (WDNR 2001). The ODFW data was derived from a land-based survey cataloguing the presence of shoreline structures. The structures data set was aligned with shoreline segments describing beach type, and a calculation was done to determine the percentage of structure across a segment. The two state data sets were merged to create a single file that indicated whether the shoreline structure was patchy (<50% coverage across a single shoreline segment) or continuous (>50% coverage).

Land cover

We utilized the NOAA Coastal Services Center data on land cover from their Coastal Change Analysis Program (NOAA CSC 2000 - http://www.csc.noaa.gov/crs/lca/ccap.html). They classified 30-meter resolution Landsat Thematic Mapper and Landsat Enhanced Thematic Mapper satellite imagery into 22 classes. We adopted the two urban classes (high and low intensity developed areas) and a cultivated class to determine levels of industrial agriculture. We also merged two additional classes (bare land and scrub/shrub) into an early seral stage class. This second category allowed us to make an initial attempt at identifying recently cut forests along the coastal range of Oregon and Washington.

Road density

Road data was collected separately for each jurisdiction within the ecoregion. All roads were treated equally in this assessment, from major highways to logging roads. This was partly because much of the base data didn't include information on road types, but more importantly because all roads have an impact on their immediate and downstream environments. Major highways may have greater impacts than logging roads, but the density of logging roads greatly outweighed all other roads types. For Washington the

POCA Transportation layer was provided by WDFW. Oregon road data was obtained from the Oregon State GIS Service Center.

Selecting High Priority Conservation Areas

One of the primary tools used in selecting areas that deserve conservation attention is the use of decision support tools. For this ecoregional study we used MARXAN, an optimal site selection algorithm (Andelman et al. 1999, Possingham et al. 2000). Decision support tools such as MARXAN help create an efficient conservation portfolio by minimizing the total area selected while meeting assigned conservation goals. Siting algorithms provide a context for objective representation that is both measurable and spatially explicit.

Decision support systems are becoming well established in conservation planning circles. Tools such as MARXAN are best suited to the situation where a study area has been divided into a set of candidate sites, or planning units, that completely fill the region (i.e., watersheds, hexagons, grid units). These are the basic building blocks for assembling a conservation portfolio. Our 520 targets were appended to a single analysis unit, a 500-hectare hexagon overlaid across the entire study area.

At the core of siting algorithms is the overall objective of minimizing the area encompassed with the network of potential reserves while meeting the desired amount of target representation (see Pressey et al. 1993). MARXAN uses a simulated annealing algorithm to evaluate multiple alternatives in site selection, comparing a very large number to identify a good solution. This function is a nonlinear combination of the total area and the boundary length of perimeter of the site selection output (Leslie et al. 2003). In its iterative nature, if a change minimizes total area relative to boundary length then the new selected set of units is carried forward to the next iteration until the maximum number of iterations is reached. It is worth noting that there is never just one optimal solution (i.e., the definitive set of conservation areas) but it is possible to identify those areas that are both essential and representative as a core part of a potential reserve system.

A scenario in MARXAN is a user-defined set of parameters including the selection of the specific type of heuristic algorithm, the desired number of solutions or runs, simulated annealing settings including the desired number iterations per run, and the amount of boundary length or clumping of planning units. In our analysis we ran the algorithm 100 times at 10 million iterations each using the simulated annealing algorithm. We also adjusted the boundary length to have a minimal effect in the solutions (set to 0.01) therefore favoring the minimization of area over shared boundary length between planning units.

Scenario 1: Site Selection without the Influence of Threats

In order to clearly evaluate the role a threats index plays in setting coastal marine priorities we first looked at how site selection functions without including an analysis of threats. Here we used an approach we call the “equal value index,” where every planning unit in the index is given the same value. We initially gave every unit a default value of 1 and compared this to a uniform scaling factor that served as a baseline for the addition of threat values. The scaling factor we used was a value of 500, which relates to the size of the hexagon planning unit in hectares. Surprisingly the outputs from the un-scaled (1) and scaled factors (500) were quite different. We assumed this was because there is a direct relationship between the area of the planning unit and the scale of values being balanced in the algorithm's function of minimizing total area. This approach provides a comparable scenario from which to assess changes in both spatial variability and efficiency in the MARXAN outcomes.

Scenario 2: Classifying and Ranking Threats
Determining actual weights to assign to each threat factor, and the relative weights within each threat, required a review of the scientific literature considering both the geographic extent and community-wide impact of the factor across intertidal and shallow subtidal communities. Not surprisingly we could not find any information on assigning specific weights to individual factors, but rather used this review to verify the inclusion of these factors in the index.

We adopted two approaches to calculating the impact of shoreline armoring, land cover classes, and road density. The first was called the "relative value index." Threats were measured for their extent of impact (i.e., the length of armored shoreline) then classified and ranked into relative weights. This method has been widely adopted in conservation planning circles, where arbitrary cutoffs are assigned within and among threats and presumed to reflect differences biologically. We include this method here to examine whether results are different from perhaps a more logical method where threats are given values based only on threat occurrence (alternative approach presented in the next section). We chose, however, not to value threats relative to each other but rather only the values within each threat. Our intention here is to place an emphasis on the factors themselves and not the individual weights, and to compare this approach with others.

We examined multiple scenarios where the weights within each threat were varied. Since there is no substantial literature to support any one assignment of relative weights, we determined them visually based on their influence over the algorithm in places where we were familiar with specific geographic conditions (Figure 4.5.3). The weight was multiplied by the base value (500) to yield an additional score, which was then added to the base.

We set two thresholds of 50% for the length of shoreline armoring that occurred both within a single shoreline segment and across any individual planning unit. The first 50% threshold classified shoreline segments with more or less than 50% structure. Segments were considered continuous if there was more than 50% structure; segments were considered patchy if they were less than 50% structure. The second threshold was applied to the percentage of continuous or patchy segments within each planning unit. Continuous segments that covered more than 50% of the total length of shoreline in that planning unit were given the highest weight. This was followed by patchy segments that represented more than 50% of the total length in a unit. The lower rankings were given to continuous or patchy segments that represented less than 50% across the total length of shoreline in a planning unit.

Using the NOAA Coastal Services Center data on coastal land cover, we extracted classes deemed to represent the most impact to terrestrial, freshwater, and marine environments. These were ranked according to the degree of land conversion and the ability for lands to be restored, from highly developed urban areas to early seral stage. This presumes that the restoration time needed to convert urban areas back to their pre-existing natural state is substantially more than agricultural and immature tree stands. These relative weights were then multiplied by the percent of that class in every planning unit. Finally, each converted land score per land cover class was aggregated to yield a single value for every unit.

The density of roads was calculated by first converting road lines to grid cells in GIS. The density of lines was calculated by evaluating "neighborhoods" of cells within which the number of cells that contained roads determined the relative density. The density output was measured in length of lines per unit area, in this case meters per square kilometer. Each cell was then multiplied by the total number of cells with that value for every planning unit. Once an aggregate number was calculated for every cell in each planning unit, these numbers were classified and assigned a value according to the range in road density. The highest densities were found in the Portland, Oregon metropolitan area and throughout Southwestern Washington.

If a particular place contained all three threats of the highest relative values, the score for that planning unit would be 1,250 including the base score of 500. The final range of values for the relative value index was 500 to 1,145 (Figure 4.5.4a). We assume that these threats represent the current condition of the upland and coastal environment and that the three factors have equal impact relative to one another.

Scenario 3: Measuring the Occurrence of Threats as Index Values

The relative value index begins with a set of threats data and the actual metric of that threat. The data for each threat is then classified and ranked according to the degree of impact across all planning units. The third scenario in our study reports on an "absolute value index." This absolute index refers to the threat occurrence values of the metric (i.e., occurrence of shoreline armoring where the metric is length) used for each factor. Whereas relative indices consider multiple factors that are all individually weighted then combined with the other factors, absolute indices preserve the occurrence value for each metric as a percentage or as the original dimensions that describe the metric. These metrics are often normalized so that each threat can be compared equally (Figure 4.5.5).

In the case of shoreline armoring, we combined the presence of all continuous or patchy segments and used the percent structured length across the planning unit as the range of values. This yielded values from zero to one. These values were multiplied by the base score, 500.

The NOAA Coastal Services Center land cover classes that contained the identified threats (high and low intensity development, cultivated lands, and early seral stage derived from bare land and scrub/shrub classes) were combined to get a total amount of hectares in each unit. This aggregated threat area was then divided by the total amount of land cover for all classes within each unit. This range also yielded values from zero to one, with one representing 100% coverage of threats.

The density of roads was calculated in the same manner as for the relative value index, where road lines were converted to grid cells and the density of lines was calculated and measured in length of lines per unit area. Each cell was then multiplied by the total number of cells with that value for every planning unit. aAn aggregate number was calculated for every cell in each planning unit and these numbers were normalized from zero to one, dividing by to highest aggregated number.

If a particular place contained all three threats of the highest absolute value, then the score for that planning unit would be 2,000 including the base score of 500. The final range of values for the absolute value index was 500 to 1,633 (Figure 4.5.4b). Like the relative index, this approach assumes that these threats represent the current condition of the upland and coastal environment. This approach, however, attempts to be more objective in its assigning of threat values. By removing the ranking of individual threats we are reporting on the actual values of each factor as it occurs on the land or seascape. This removes much of the debate among planners and reviewers as to the method for classifying and ranking factors relative to each other.

Measuring Solutions

There are two main outputs from MARXAN, including a "summed solution" that tracks the number of times a unit is selected across all runs in a scenario, and a "best solution" which represents the most optimal single solution in a scenario. Keeping track of how often each unit was involved in any solution is a useful way to explore the relative irreplaceability of units across the study area. This output, as well as the best solution, is quite useful during subsequent expert reviews in determining high priority conservation sites. We compared solutions from the three different scenarios to examine 1) the spatial variability of the algorithm choosing a particular planning unit, and 2) the spatial efficiency in meeting conservation goals from a single scenario.

The spatial variability here is defined as the number of times a particular planning unit was chosen by the algorithm across multiple solutions in a single scenario. This is a measure calculated from the summed solution MARXAN output described above. The first measure was to calculate the number of units chosen at least once across 100 runs (from a total of 9,912 units). This gave us an overall determination of the variability in selecting sites across a large geography. Our second spatial variability measure was to divide the 100 summed solutions into 5 ranges of equal interval. The lowest 20% selected (1-21 times) was used to determine the relative amount of variability of each scenario. For instance, if the number of units represented in this range was relatively higher when compared to those in another scenario, then we determined the overall output to be highly variable. This is in contrast to the highest 20% selected (81-100 times). Units in this range were used to examine the level of "irreplaceability" of the scenario. If the number of units selected here was relatively higher when compared with units in another scenario, then we determined that the scenario was less variable in its overall selection, choosing more irreplaceable or core areas. The other 20% ranges (20-40%, or 22 to 41 times selected; 40-60% or 42 to 60 times; and 60-80% or 61 to 80 times) did not contain significant differences across scenarios and therefore we did not utilize them as measures.

The spatial efficiency in meeting conservation goals was measured by 1) the number of units selected as the best solution and 2) determining the number of conservation targets that over-represented assigned goals, precisely met goals, adequately met goals, or did not meet them. We adopted and modified three terms or categories (Leslie 2003, Ferdaña 2005) to quantitatively describe and examine spatial efficiency: overrepresentation of targets (>130%), efficiently captured targets (p = < 130% and > 97%), and targets not met (< 97%). We also looked at the total number of units selected across the study area to determine the efficiency of selecting high priority conservation areas. Both of these measures were examined by utilizing the best solution from MARXAN.

To test the variability and efficiency question we ran each scenario twice with the exact same parameters. This was done to determine the percent variation of the algorithm for the same scenario. For spatial variability we found a variation between 0.1% and 0.9% in the number of units selected at least once in 100 summed solutions across scenarios. We therefore considered a difference of more than 0.9% between scenarios to be statistically significant. For spatial efficiency we found a 1 to 7 difference in the number units selected in best solution across the 3 scenarios. This represents a variation between 0.1% and 0.4% in the number of units selected in best solutions across scenarios. Therefore we considered any difference over 0.4% in best solution statistically significant. For the purposes of simplicity we rounded our results to the nearest tenth of a percent.

Since this is an integrated analysis we evaluated the variability and efficiency questions separately and combined among the terrestrial and marine environments to test whether they reacted differently across scenarios. For spatial variability we found that separating out land and sea units did not yield different results. Therefore we combined them in examining the multiple scenarios. For spatial efficiency, however, we found that there were significant differences among scenarios in both environments. Therefore we will report here on spatial variability as a combined land-sea analysis, and spatial efficiency as a comparison of the number of terrestrial and marine targets captured in each scenario.

4.3 Results

Scenario 1: Equal Value Index

Using the base value of 500 applied to every planning unit, the no threat or equal value index yielded 6,416 units, or 65% of the total number of units, that were chosen at least once in 100 runs of the algorithm. This reflected high spatial variability overall, where well over half the total number of units were part of at least one solution. The algorithm was not constrained in its selection of units, but was driven largely by capturing variations of representative land-sea targets. In examining the lowest 20% range from the summed solution we also determined that there was high spatial variability compared with the other scenarios (Figure 4.5.6). There were 4,169 units represented in this range, or 65%. In looking at the level of irreplaceability in this scenario there were 797 units selected in the highest 20% range, or 12%. This also illustrates the high spatial variability of the scenario and its inability to capture more irreplaceable units (Figure 4.5.7a). The algorithm had a lot of choice in its selection of units which is an indication that impacted terrestrial and marine conservation targets did not influence the results.

In terms of spatial efficiency, the best solution chose 1,696 planning units, or 848,000 hectares. This represents 17% of the region selected, or 353 marine and 1,343 land units. We then utilized the three target categories (overrepresentation, efficiently captured, not met) and applied to both terrestrial and marine targets (Figure 4.5.8). For terrestrial targets, 33% were found to be over-represented, 48% were efficiently captured, and 19% were not met. Among the estuarine and nearshore marine targets 59% were over-represented, 20% were efficiently captured, and 21% were not met. From this analysis the terrestrial targets were more efficiently captured in the scenario than the marine selection. In addition the terrestrial targets were not nearly as over-represented as marine targets (33% versus 59%); both were similar in the number of targets not met.

Scenario 2: Relative Value Index

Applying the relative value index to MARXAN, the algorithm chose 5,603 units at least once in 100 runs (57%). This is 8% fewer than that of the equal value index, indicating that the relative weights constrained the algorithm's selection of units across the region. Evaluating the lowest 20% range of summed solution there were 3,272 units selected, or 58% (Figure 4.5.6). These numbers also illustrate that the algorithm more efficiently optimized results in the relative index compared to results using the equal value index in that fewer units were selected in the lowest range (difference of 7%). For the most irreplaceable, or the highest 20% range, 879 units were selected, or 16%. More units were considered core or irreplaceable to the selection output (a 4% increase) when applying terrestrial and marine-based threats (Figure 4.5.7b).

In terms of spatial efficiency, the best solution chose 1,702 planning units, or 851,000 hectares. This also represents approximately 17% of the region selected, or 355 marine and 1,347 land units. These differences among scenarios are insignificant, revealing the fact that the efficiency of the best solution does not increase when regional terrestrial and marine-based threats are applied. We also found similar results when comparing the amount of terrestrial and marine targets meeting goals in analyses with a relative index as compared to the equal value index (Figure 4.5.8). The results with the relative index varied from the equal value index scenario in several ways. First, the terrestrial targets were less efficiently captured in this scenario (46%), down 2%. Second, the marine target selection was more efficient by 2% (22%). In terms of over-representation the terrestrial targets were 35% over-represented, up 2% from the equal value index scenario. For the marine targets there was less over-representation, here down 3%. Though the 2 to 3% decreases and 2% increases were not considered significant in and of themselves, they represent a shift in the algorithm to a slightly more efficient marine selection when threats were applied.

Comparing the terrestrial with the marine output, the terrestrial selection was more efficient in reducing over-representation (35% versus 56%) and meeting goals (46% versus 22%). Like the equal value index scenario, both the terrestrial and marine selection was similar in the number of targets not met, 19% and 22% respectively.

Scenario 3: Absolute Value Index
Using the absolute value index there were 5,400 units selected at least once (55%). These percentages are strikingly similar to results with relative value index, though the number of units chosen more than once was down 2%. This reflects the fact that the absolute value index contained a larger range of values which further constrained the algorithm to select specific units. In examining the summed solution ranges, there were 3,081 units or 57% found in the lowest 20% range (Figure 4.5.6). Again these numbers are quite similar to results with the relative value index but both illustrate less spatial variability than the equal value index. The highest 20% range, considered most irreplaceable, contained 17% of all units. This reinforces the notion that both the relative and absolute scenarios reduced the overall spatial variability while increasing the number of irreplaceable sites (Figure 4.5.7c).

For spatial efficiency, the best solution chose 1,703 planning units, or 851,500 hectares. Similar to the other two scenarios, this represents approximately 17% of the region selected, or 354 marine and 1,345 land units. In the evaluation of target representation we also found similar results when comparing terrestrial and marine targets to the relative value index scenario (Figure 4.5.8). In the terrestrial environment 35% were found to be over-represented, 46% were efficiently captured, and 19% of targets were not met. Among the estuarine and nearshore marine targets 57% were over-represented, 23% were efficiently captured, and 20% were not met. This scenario was so closely correlated with the relative value index scenario that it was not statistically distinct. It does, however, also make the same shift away from results using the equal value index in having a more efficient marine selection and a less efficient terrestrial one.

4.4 Discussion

This study has been an evaluation of the influence that information on threats has on regional conservation planning. We have shown there are numerous threats that affect priority setting in terrestrial and marine environments. Including threats influences site selection analyses by forcing the algorithm to choose fewer sites which are more consistently selected. There is less variability in the result since the algorithm avoids areas with less desirable conditions or a higher degree of threat. Therefore using the summed solution output of MARXAN as a measure for spatial variability is a useful indictor of the influence threats have on the selection process.

The influence of threats on spatial efficiency, however, was much less significant. Differences among scenarios were insignificant, especially when comparing the results using the relative and absolute indices. The algorithm was able to meet goals consistently in all three cases as well as have a similar amount of overrepresented targets. Therefore using the number of best solution units selected regionally and comparing the target categories is not a good measure of spatial efficiency.

Selection results using single or multiple threat factors have been illustrated quantitatively, allowing comparisons between indices. Cumulative effects from multiple threats present in a location or over a large geographic extent can have community-wide impacts. The influence of threats is most evident when examining the separate terrestrial and marine best solution outputs. Overall the terrestrial selection was more efficient. This was largely due to the fact that the terrestrial input data was of a similar scale to the planning unit (500 hectares), where marine shoreline targets tended to be excessively aggregated into single units along the coast and thereby overrepresented. When threats were applied to the terrestrial selection, however, impacts significantly altered results and led to a less efficient selection. This makes intuitive sense in that land use practices and the density of roads affects the distribution of species and impacts their habitat. The algorithm effectively took threats from logging, agriculture, urban areas and roads into account relative to the impacts of shoreline armoring regionally. The marine selection, however, improved in efficiency when shoreline armoring was applied. The marine information lacked viability rankings in all cases, and therefore the selection process was based largely on quantitative representation. There were more choices for the algorithm in meeting target goals. This was not the case for terrestrial, where specific plant targets contained individual viability ranks and therefore caused the algorithm to select targets even within impacted areas. This also caused the terrestrial selection to be less efficient.

From this study we believe it is important to conduct sensitivity analyses when combining multiple threat factors. Our analysis examined three approaches to constructing a threat index, and there are others (Floberg et al. 2004). We have demonstrated that using an index where values are equal helps measure the influence of weighted indices. For instance, when comparing the equal with the relative index, more units were contained in the highest summed solution range (81-100%) when applying the relative weights. For conservation planners the selection of places in this range helps focus in on core or irreplaceable sites. Developing different weighted indices may help determine the appropriate approach for a specific geography, or reveal that more than one method may be suitable. From this analysis we found that adding a threat index to site selection is important in order to both quantify the influence of combined factors and evaluate the ecological integrity across large land and seascapes. Comparing approaches to developing a threat index will lead to more credible reporting on the influence of those threats and their cumulative effects. We conclude that although the different approaches applied here resulted in similar results, either weighted index was significant when compared to not including these factors in regional conservation planning efforts. With this as an initial examination, we suggest using this approach as a baseline to conduct similar sensitivity analyses, including a scenario where individual factors are more heavily weighted. This will be beneficial for examining worst case scenarios and how the algorithm reacts to a dramatically impacted environment. Through this process we believe users of the decision support tool will be able to better calibrate the index to suit real world geographic conditions.

Using a decision support tool such as MARXAN will allow planners to test their assumptions about terrestrial and marine-based threats by drawing on quantifiable measures from the algorithm's various solutions. Through this process more informative management decisions can be made that will help guide the implementation of appropriate conservation at high priority sites. There are a number of other measures that could be used to test the spatial variability and efficiency of site selection. For instance, spatial variability can be characterized as the shifting of planning units across scenarios as the number or weight of threats changes. A preliminary look using the best solution output revealed that units shift significantly even when the same scenario was run twice. Using our equal value index scenario, we found only a 66% overlap of best solution units when ran twice with the same parameters. Likewise for the relative and absolute index scenarios, we found only a 72% overlap. This indicates that there is a level of inherent variability of units being selected when running the same scenario multiple times. This geographic-based variability has significant implications when testing the sensitivity of the algorithm and quantifying statistically significant results from its solutions.

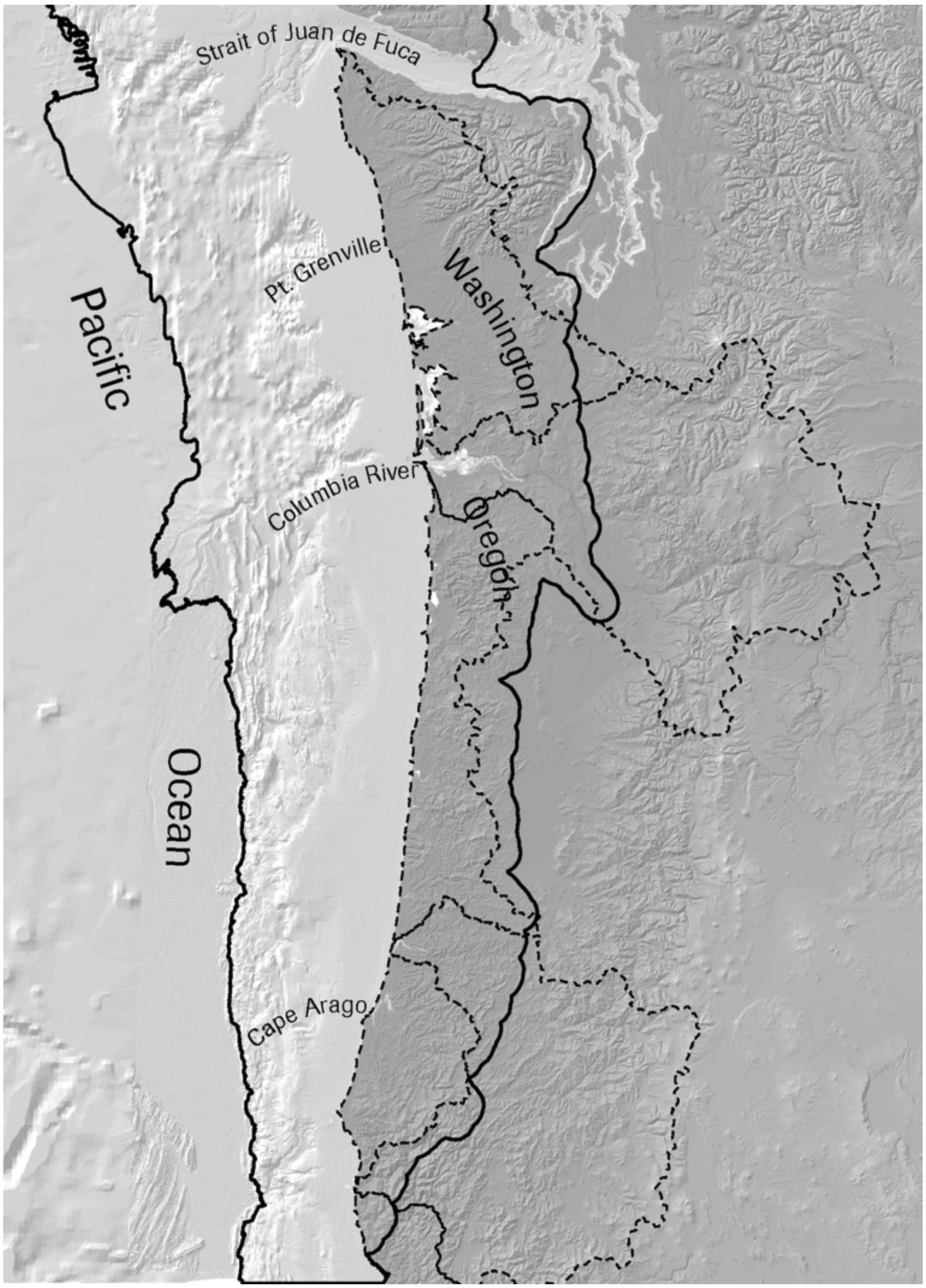
We also compared the variability of best solution between the equal and relative index scenarios as a method of quantifying the reduction of threats. For example, we calculated 1,800 kilometers of shoreline structures (e.g., sea walls) along the Oregon and Washington study area. Using the equal index scenario best solution captured 591 kilometers of structure in order to meet assigned conservation goals. Employing the relative index scenario 551 kilometers were captured in the solution, a reduction of 40 kilometers or 7% in the solution. We also looked at the length of roads to illustrate this point. The total length of all roads in the area was calculated at 105,564 kilometers. In the best solution for equal index there were 12,651 kilometers of roads within the selected units. Adding the relative index to the selection process we captured 9,997 kilometers of road in the solution, a reduction of 2,654 kilometers or 21%. This reduction of threats in the solution aids in the targeting of biological features that are not in the same geography as these impacts, which may help planners and managers identify areas of relative ecological integrity.

It is important to note that suitability indices often use information on managed areas as a way to account for what resources are contained within them, and offset the influence of threats in the selection process. Although we did not use marine managed areas in this analysis the algorithm did chose areas along the Olympic coast that are currently under a variety of protective designations (i.e., National Park Service, National Wildlife Refuge, National Marine Sanctuary). This is a result of the unique biological and physical features in this geography, but may also be an indication of management effectiveness in protecting species, habitats and ecosystems as well as abating current threats. The relationship of managed areas and threats needs more examination. As Stoms (2000) notes, the impacts such as high road density can occur despite the fact that they are located within moderately well protected areas. In addition, there are a number of terrestrial parks that have marine jurisdiction along coastal environments, although often this is hard to decipher from available information sources. More analysis of the role managed areas play in priority setting in this geography needs to be conducted.

Our analysis demonstrates that land-based threats have a significant effect on terrestrial site selection. The same is true for evaluating the nearshore marine environment. More study, however, is required in order to accurately measure the influence of threats at the land/sea margin. We believe that terrestrial and marine site selection is directly correlated to the number and level of influence given to both land and marine-based threats. Although we showed that including more factors will continue to influence spatial variability and efficiency where the impact or cumulative impacts occur, there also needs to be a quantifiable method for determining the effects of land-based threats on the nearshore marine environment when conducting an integrated analysis. As watershed processes influence the condition of estuarine and coastal processes, planners need to account for this connectivity through integrated planning efforts. Such factors might include percent logged or impervious surface in a watershed, or the number of invasive species in estuaries. By including these factors it is possible to calculate their influence at the stream mouth and carry that impact out into the coastal zone using both freshwater flow and nearshore current parameters. We conclude that while this examination helps us evaluate the influence of threats in land and sea environments, more work is required to truly determine conservation priorities at the land/sea margin as well as in terrestrial and marine environments.

4.5 Figures

4.5.1 The U.S. portion of the Pacific Northwest Coast ecoregion. Thinner, dashed lines illustrate freshwater ecological drainage units within the ecoregion.



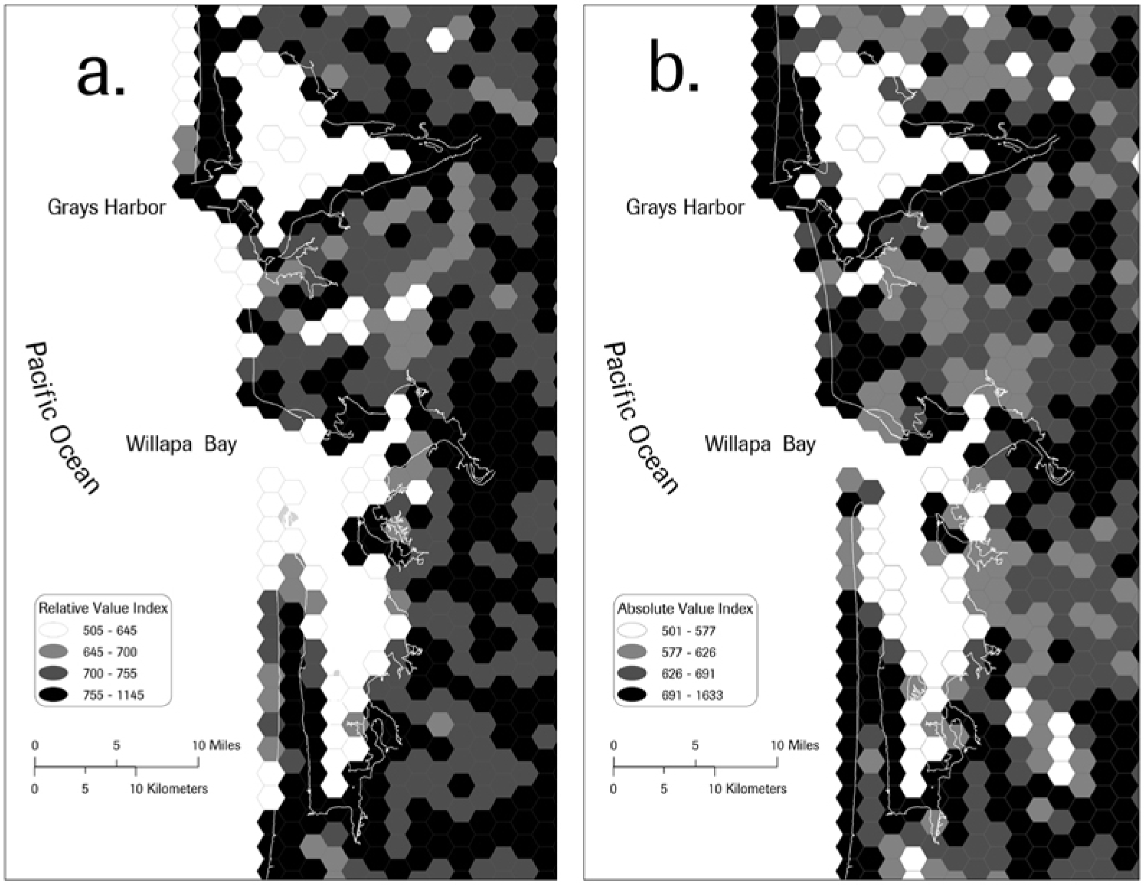
4.5.2 Spatially-explicit threats used in the analysis

Threat	Zone of influence	Effects	Marine targets impacted	References
Shoreline armoring	Intertidal and shallow subtidal	Scouring effect by backwash from wave action Loss of coastal habitat Contributes to anthropogenic, chemical inputs	Meiofaunal abundance in estuarine sand beaches Fishes and crustaceans Zoo- and ichthyoplankton	Peterson 2000; Spalding 2001; Gordina 2001; Galbraith 2002
Land use: coastal development, cultivated lands, early seral forsts / logging	Terrestrial, marine riparian, estuarine, nearshore	Alters the physical condition of the shoreline Alteration of sediment processes Decline of coastal water quality Nutrification of aquatics environment Chemical inputs from logs and industry Surficial erosion Lost topsoil, siltation and burial of aquatic life	Fish Shellfish Sea and shorebirds Shoreline habitats Seagrasses Estuarine soft bottoms	Shreffler et al. 1994; Doyle et al. 2001; Beach 2002; PEW Oceans Commission 2003; U.S. Commission on Ocean Policy 2004
Road density	Terrestrial, marine riparian, estuarine, nearshore	Runoff of chemcials and sediment to streams Erosion	Aquatic ecosystems	Stoms 2000; Angermeier et al. 2005; Ziegler et al. 2005

4.5.3 The three threat factors and their scores included in the relative value index.

Threats	Weight	Method
Shoreline armoring		
Structure > 50% across shoreline segment - continuous	0.5	> 50% structured length \ total length across unit
Structure < 50% across shoreline segment – patchy	0.4	> 50% structured length \ total length across unit
Structure > 50% across shoreline segment – continuous	0.3	< 50% structured length \ total length across unit
Structure < 50% across shoreline segment - patchy	0.2	< 50%structured length \ total length across unit
Land cover		
High intensity developed	0.5	weight times % area in unit
Low intensity developed	0.4	weight times % area in unit
Cultivated land	0.3	weight times % area in unit
Early seral stage (bare land and scrub/shrub)	0.2	weight times % area in unit
Road density		
Road density (81 – 100% quantile)	0.5	meters\square km times number of cells in unit
Road density (61 – 80% quantile)	0.4	meters\square km times number of cells in unit
Road density (41 – 60% quantile)	0.3	meters\square km times number of cells in unit
Road density (21 – 40% quantile)	0.2	meters\square km times number of cells in unit

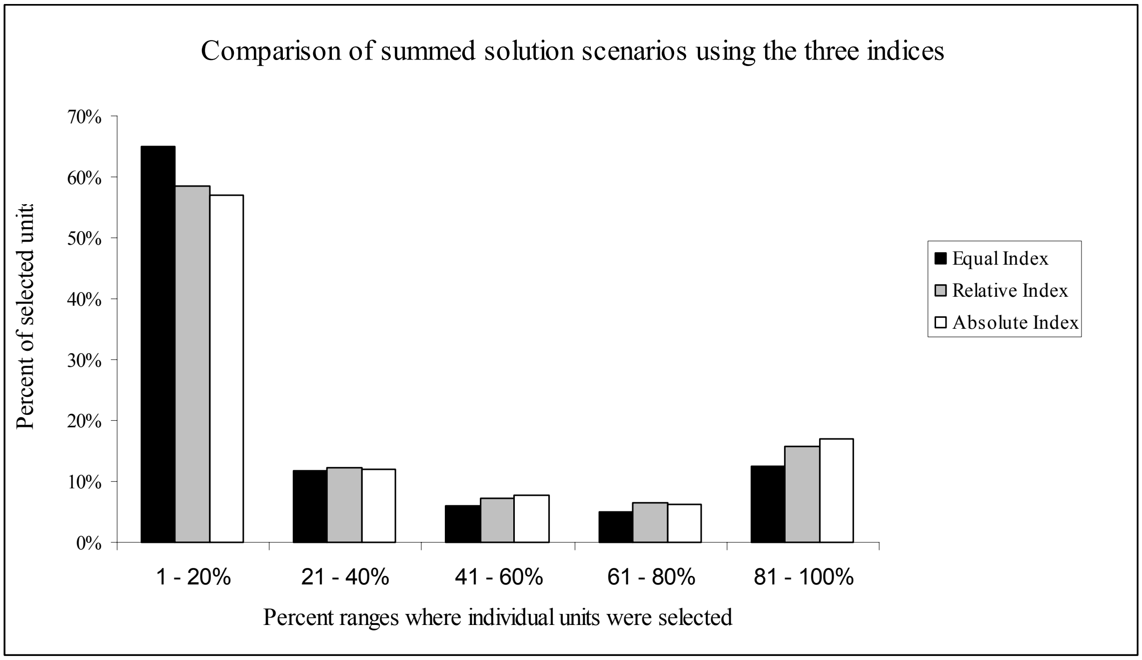
4.5.4 The two threat indices: a) relative and b) absolute index (southern Washington)



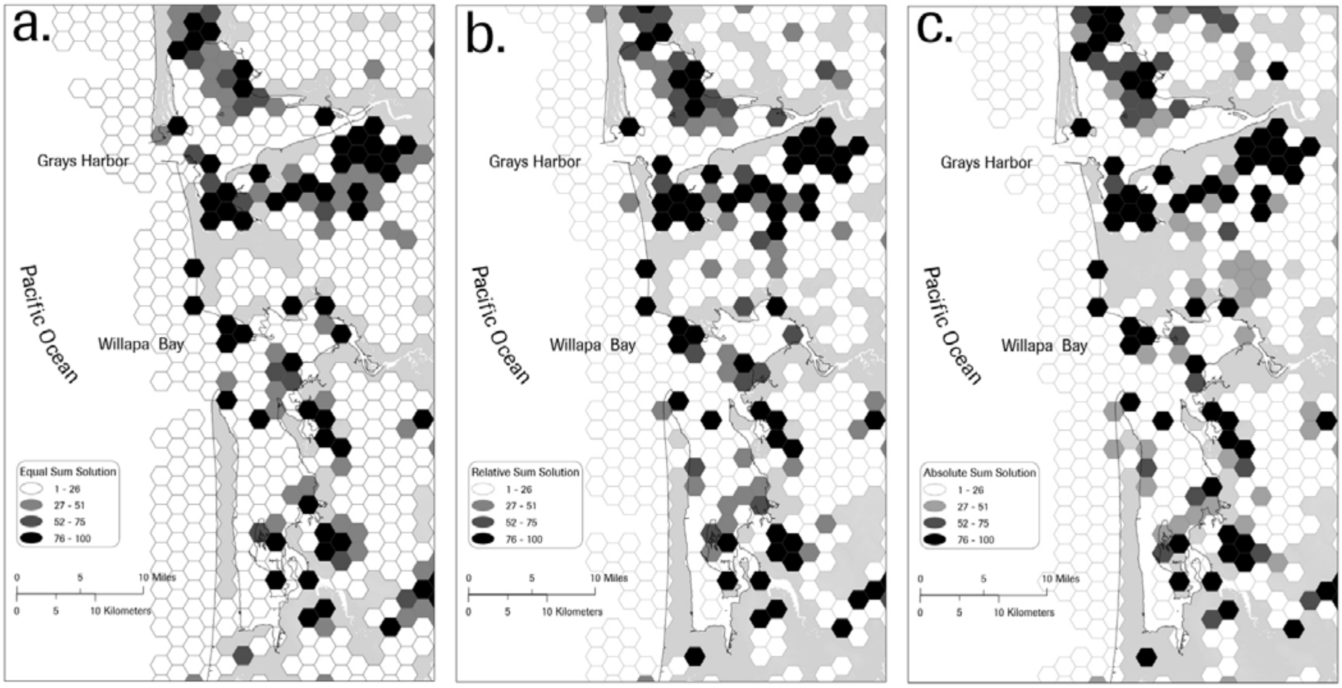
4.5.5 The three threat factors and their scores included in the absolute value index.

Threats	Range	Method
Shoreline armoring		
All continuous and patchy shoreline segments	0 - 1	% structured length \ total length across unit (meters)
Land cover		
All land cover classes with threats combined	0 - 1	% of all land cover threat classes \ total area of unit (hectares)
Road density		
All road density values	0 – 1 normalized	meters\square km times number of cells in unit; normalized not ranked

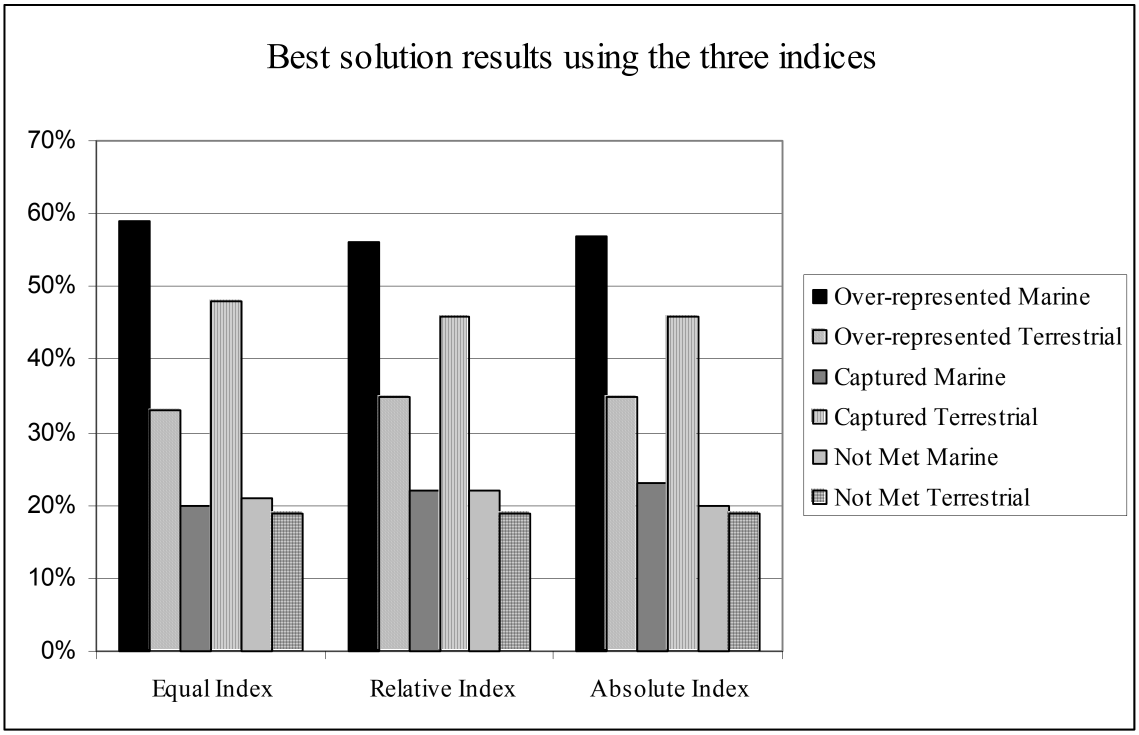
4.5.6 Bar chart illustrating the output of the MARXAN summed solutions. The output was categorized into percent ranges and each index was compared by evaluating the number of units selected at least once: the scenario using the equal index selected a total of 6,416 units, relative index selected 5,603 units and the absolute index selected 5,400 units



4.5.7 Summed solutions of the three indices: a) equal, b) relative, and c) absolute (southern Washington). The shades represent the number of times that planning units were selected in different runs. Darker colors indicate units that were selected more often.



4.5.8 Application of the three target categories to the three indices: overrepresentation of targets (>130%), efficiently captured targets (>130% and < 97%), targets not met (p = < 97%)



5.0 Integrating Conservation across Land and Sea: A Direct Comparison of Regional Planning Approaches in the Pacific Northwest, USA.

(For submission to *Frontiers in Ecology and the Environment*)

Abstract

There is a need for better integration in conservation and management across terrestrial, freshwater, and marine environments. Regional plans often serve as the basis for allocation of conservation and management effort. If these plans account for connections across environments, then the appropriate placement of resources and effort should follow. Conceptual approaches have been offered for how we can integrate terrestrial, freshwater, and marine information in to systematic planning, but specific case examples have not been tested.

To understand how to better coordinate conservation and management efforts across terrestrial, freshwater and marine environments, we examined integrated and unintegrated approaches for planning and quantitatively compared their efficiency and accuracy. We compared approaches using two regional planning efforts by The Nature Conservancy (TNC) and partners; the Puget Trough ecoregion of Oregon, Washington and British Columbia and the Cook Inlet ecoregion of Alaska. In both ecoregions, integrated analyses across environments led to clear gains in spatial efficiency over unintegrated analyses with gains much stronger in the Cook Inlet than the Puget Trough ecoregion; the gains in spatial efficiency were 36%, and 5% respectively. In the Cook Inlet ecoregion, the planning units selected in the integrated analysis (Figure 4a) tended to cluster consistently and efficiently along riverine areas from the summit to the seas. These gains in spatial efficiency are likely to result in real gains in economic efficiency (i.e., reduced costs in conservation), because resources can be focused on fewer places to meet all conservation objectives. The results were mixed for a measure of ecological accuracy; the precision with which goals were met for the biological targets. Goals were met slightly more precisely in the unintegrated analyses in the Puget Trough ecoregion, but goals were met much more precisely for integrated analyses in the Cook Inlet ecoregion.

These results demonstrate that there can be strong benefits to integrated planning with some limitations as well. Coastal planning efforts should at least examine integrated results and we offer advice on how that can be done analytically and practically.

5.1 Introduction

The boundary between land and sea is fluid and permeable. Many species and processes move freely across this boundary including humans and their impacts. We know that there are important connections between terrestrial, freshwater, and marine environments in species (e.g., seabirds) and threats (e.g., nutrients, oil spills, urban sprawl) and in the strategies to address them (Beck 2003; Beck et al. 2004). Nonetheless science, conservation, and management are often artificially divided across the boundaries of land, river and sea, which results in fractured conservation and management in the coastal zone. The problems created by this fractured governance were highlighted in recent reports by high level commissions (Pew Oceans Commission 2003; U.S. Commission on Ocean Policy 2004).

Given the historical divisions in science, conservation and management across the coastal zone, why should we worry about integrating efforts? To improve accuracy and efficiency! Efforts that recognize these connections across environments can be more ecologically accurate and economically efficient. Regional plans often serve as the basis for allocation of effort and if they account for connections across environments, then the appropriate placement of resources and effort could follow. We illustrate why we should try to integrate efforts better across environments, then test if and how systematic planning could help in integrating efforts by comparing integrated and unintegrated regional planning efforts in the Pacific Northwest.

Why Integrate? Improve Ecological Accuracy

The first set of reasons for better integration in efforts is that it is logical, i.e., eco-logical, to understand, conserve and manage the many connections across environments in species, ecosystems, and processes. There are real connections between environments and incorporating these connections can improve resource management.

Ecosystems that straddle environments cannot be considered to be terrestrial, freshwater or marine and require an understanding of processes, fluxes and connections across environments. Indeed all estuarine ecosystems must be understood, by definition, to be partly in freshwater and partly in marine environments. The same is true for all intertidal environments, which are partly marine and partly terrestrial environments.

Many species have life histories that require physical connectivity and passage between the different environments. Their populations cannot be managed appropriately if these connections are not conserved and maintained. These species primarily fall in to two categories, those with joint marine and freshwater requirements (e.g., anadromous and catadromous species including salmonids, sturgeon and eels) and those with joint marine and terrestrial requirements (marine feeders/terrestrial breeders including seabirds, seals, sea lions, and turtles).

We make ecological mistakes that may result in inefficient or even wasteful conservation and management when we do not account for the full range of the life history of these species. For example, it is popular strategy for sea turtle conservation to protect turtle nesting grounds (e.g., night patrols and removals of eggs for artificial rearing). These activities are time and cost intensive. Crouse et al. (1987) have shown for loggerhead turtles in the Gulf of Mexico that the critical life history stage for mortality is in the transition from the juvenile to adult stage in males. While efforts to protect nesting loggerhead mothers and their eggs may be inspiring and feel good, if we want to ensure the viability of these populations, then management needs to focus on the protection of juvenile males to ensure that they reach adulthood.

Some marine planning and management efforts recognize connections across environments usually because of obvious impacts to the marine environment from upstream sources. Some of the most obvious upstream effects that provide connections among environments are related to freshwater inflow including the rate, magnitude, and timing of input of fresh water, sediments, nutrients, and pollutants (Wilber 1992; Turner & Rabalais 1994; Wilber 1994; Loneragan & Bunn 1999; Mitsch et al. 2001). Development within watersheds also has many implications for coastal environments. Evidence suggests that when more than 30% of a watershed is developed with impervious surfaces that there are substantial effects on near-shore biological communities. Even 10% development of impervious surfaces in a watershed can lead to significant negative effects on coastal marine systems (Beach 2003).

It is less common for freshwater and terrestrial efforts to consider connections that arise downstream, but these connections do go both ways. Many coastal species from spiders to bears receive sustenance from the seas. Terrestrial planning efforts that do not consider factors such as effects of overfishing on seabirds or estuarine concentrations of pollutants that rise through coastal food webs will not accurately identify appropriate conservation actions for terrestrial species and ecosystems. For example, Helfield and Naiman (2001) found that 22-24% of the nitrogen in trees and shrubs near salmon spawning streams was marine derived. This marine subsidy significantly influenced growth rates in Sitka spruce trees (*Picea sitchensis*), which may in turn make these streams better for salmon spawning, because of increases in shading, sediment and nutrient filtration, and the production of large woody debris. If the source of these nutrients and the connections among environments is not recognized and managed, we may begin to lose the forest for not seeing the fish.

Why Integrate? Improve Economic Efficiency

The second set of reasons for integrated conservation and management is that it makes good economic sense for efficiencies in planning and management (e.g., the development of preserves, parks, and programs). The need for efficiency and integration of effort across environments is greater in the coastal zone than anywhere else, because demand for these environments is high and conservation and management in these environments is more expensive than elsewhere (Dobson et al. 1997). Coastal environments and estuaries are where the majority of people have chosen to live, work, and play for centuries. The coastal zone contains some of the most heavily impacted marine environments (Edgar et al. 2000) and the highest concentrations of rare and imperiled terrestrial birds and mammals.

There is efficiency to be gained in integrating efforts across environments. For example, it is cost efficient to do one integrated plan instead of three separate efforts across terrestrial, freshwater and marine environments. Integrated planning can result in improved communication and fewer redundancies in effort (Beck 2003, Ferdaña 2005). For example, many of the threats considered by planners are similar across environments and this information could be compiled just once instead of three separate times.

Integrated conservation and management can lead to more efficient investments in programs. Many areas with high estuarine and near shore marine diversity and productivity occur in areas where uplands are intact and diverse. This correlation occurs in part, because many stresses in coastal waters arise upstream. In many places even if there are not strong direct connections, there appear to be correlations among terrestrial and marine hotspots in biodiversity (Roberts et al. 2002). There is a clear efficiency to co-locating investments in programs and offices across environments for working with partners and in staffing for monitoring and enforcement.

Moreover, conservation impact is enhanced when actions can be clearly demonstrated to affect species and ecosystems across environments. When actions have benefits across multiple environments and jurisdictions, there can be dramatic increases in the congruence, cooperation and support among groups and agencies. For example, organizations focused on improving practices on agricultural lands find new support for these actions when they can be shown to benefit swimming and fishing downstream.

Planning for Better Integration

One way to achieve better integration of conservation and management across the coastal zone may be to develop plans that account for these ecological and economic connections among environments. By establishing shared processes and information resources, we may be able to move towards decision-making processes that are more accurate and efficient.

Increasingly scientists, agencies and organizations are using systematic planning approaches to identify where and how to allocate effort particularly at regional levels (e.g., Possingham et al. 1999; Day & Roff 2000; e.g., Beck & Odaya 2001; Leslie et al. 2002; Airame et al. 2003). These approaches enable decision makers to develop a range of solutions to resource planning and to examine how changes in decision can affect priorities. These approaches are adaptable, transparent, and repeatable.

There are three types of integration possible in planning; unintegrated, partly integrated and integrated plans. In fully integrated planning efforts, terrestrial, freshwater and marine environments are considered jointly in one plan. For example, in planning efforts that use site selection algorithms, information is included across all environments as part of one analysis or planning effort. By unintegrated, we mean plans done entirely separately in two or more environments.

There are a whole range of planning efforts that could be considered partly integrated where the plans are mainly done independently across environments but recognize some connections in species, ecosystems, and processes at some point in the process. Even when analyses are done entirely separately in each environment, some integration can still be achieved through the identification of processes and threats that have direct connections across environments. For example, factors that are indicative of degraded connections among environments include shoreline hardening, input of upstream pollutants and nutrients, and % impervious surfaces in watersheds. Efforts can also be partly integrated by comparing priority areas across environments. Even when planning efforts are run separately in different environments, the results can be compared across environments and it may be possible to adjust the location of priority sites before efforts are finalized.

We offer advice and examples towards more integrated planning. Some conceptual approaches towards greater integration have been described but have not been demonstrated with real world applications (Beck 2003; Stoms et al. 2005). We focus mainly on a direct comparison of integrated and unintegrated approaches and use real world regional examples in the Pacific Northwest to compare approaches. We offer practical advice for better integration that can be used directly in systematic planning approaches (e.g., Pressey 1993; e.g., Margules & Pressey 2000; Groves et al. 2002; Groves 2003), and most advice is germane to other planning approaches as well, whether for single species, multi-species or regional biodiversity plans.

5.2 Methods

To understand how to better coordinate conservation and management efforts across land and sea, we compared integrated and unintegrated approaches for planning and examined their efficiency and accuracy. We compared approaches using two regional planning efforts by The Nature Conservancy (TNC) and partners; the Puget Trough ecoregion of Oregon, Washington and British Columbia and the Cook Inlet ecoregion of Alaska.

Both plans followed the same basic approach for regional planning that is now commonly used by many other organizations and agencies (Beck & Odaya 2001; Groves et al. 2002; Leslie et al. 2002; Groves 2003; Leslie 2005). The basic elements of this approach are outlined in Figure 5.5.1. The basic premise is that there are a set of biological targets such as species and ecosystems in a geographic region and areas are for conservation and management, which meet stated objectives such as representation of all targets at specific goal levels with the least total area or cost.

The site-selection tools, Spexan and SITES were used to analyze the data and identify a set of sites to meet the planning objectives. Spexan uses a simulated annealing algorithm developed by Ball and Possingham (Ball & Possingham 2000; Possingham et al. 2002) to identify a representative and spatially efficient network of sites. The algorithm attempts to iteratively identify the 'best' solution to meet the objectives, which is an approximation of the unknown optimal solution. The best solution was identified from 100 separate runs (or solutions) of the algorithm with ten million iterations being examined in each run. Although an optimal solution cannot be identified, when this process was repeated with the same parameters (targets, goals, etc.) in these regions, the 'best solutions' differed by less than 0.5% in the total area identified. We used the SITES ArcView interface to map the sites selected in integrated and unintegrated approaches in a geographic information system.

The analyses presented here were used to inform these planning efforts and are interim analyses that were modified by further review with TNC staff, external scientists and stakeholders before the plans were finalized. The final results of these planning efforts are published elsewhere (Conservancy 2003; Floberg et al. 2004; Ferdaña 2005) (www.conserveonline.org/workspaces/MECA). The focus of TNC's efforts in regional planning is on the identification of priority areas for conservation; no presumption is made about the right strategies for management in those areas (e.g., restoration, protected areas, nutrient reduction). The identification of appropriate strategies occurs in more focused planning within the priority areas.

Comparing Approaches

Within each region, we compared ecological accuracy and spatial efficiency of the results of the integrated analyses with the combined results from the independent or unintegrated analyses in each environment. We used geographic or spatial efficiency as a measure of economic efficiency. Geographically, an efficient network of sites is one which is able to accomplish all conservation objectives with the least total area, because it is assumed that total area is related to total conservation cost and that smaller total areas will be cheaper to conserve than larger areas.

Ecological accuracy is much more difficult to measure and compare across approaches. As a proxy for accuracy, we compared how precisely the stated goals were met for every biological target in each approach (Leslie et al. 2002; Ferdaña 2005). The goals to be met were, for example, 20% of the current distribution of salt marshes in each region. If goals were not met, then the targets are under represented in the final set of areas. If goals were greatly exceeded, then the targets were overrepresented. Overrepresentation was less of a problem than under representation from a conservation standpoint but if goals were greatly exceeded this is certainly a less precise and less preferred plan. A goal for a target was assumed to be met if the areas included 97 - 130% of the stated goal (i.e., desired distribution) for the targets. If areas included < 97% or > 130% of the stated goals, then targets were under represented or over represented, respectively. Ultimately the true measure of ecological accuracy would require us to conserve all the areas identified in plans and determine over time if we had identified the areas that maintained the biological targets over time. The fewer targets conserved over time, the less ecologically accurate the plan was. This measure is obviously difficult to obtain.

Although the second set of analyses in each region were "unintegrated", they were informed by the same suitability index as used in the integrated analysis. That is, there was only one set of suitability factors identified for the different environments in the regions. Thus even these "unintegrated" analyses were in fact partly integrated.

Puget Trough Ecoregion, Oregon, Washington and British Columbia

The Willamette Valley-Puget Trough-Georgia Basin ecoregion is a long ribbon of broad valley lowlands and inland sea flanked by the rugged Cascade and coastal mountain ranges of British Columbia, Washington, and Oregon. It encompasses some 5,550,000 ha of Pacific inlet, coastal lowlands, islands, and intermontane lowland, and extends from the Sunshine Coast and eastern lowland of Vancouver Island along Georgia Strait, south through Puget Sound and the extensive plains and river floodplains in the Willamette Valley. Although the ecoregion's elevation (land portion) averages 445 feet (maximum 4,203 feet), the effect of the adjacent mountains, ocean intrusions, and glaciation in the region's northern two-thirds have caused dramatic localized differences in climate, soils, and geology. From distinctive combinations of these factors spring an array of systems ranging from coniferous forests to open prairies, rocky balds, and oak savannas. The marine and estuarine environments of British Columbia and Washington include systems such as kelp forests, eelgrass beds, and saltmarsh as well as many marine mammals, fishes and invertebrates.

The Puget Trough ecoregional assessment was co-sponsored by TNC and Nature Conservancy Canada (NCC) and involved partners from state/provincial and federal agencies, academia, and other non-profit organizations. Detailed methods for the overall effort can be found in Floberg (2003). Ferdana (2005) describes the marine planning parts of this effort in depth.

There were many different types of species and ecosystems (e.g., kelp forest) identified in the Puget Trough ecoregion (Figure 5.5.2). The Puget Trough region was stratified into two marine and four terrestrial subregions. After accounting for potential stratification in the region, there were a total of 108 nearshore marine targets and more than 800 terrestrial targets. Subregions are identified to account for potential geographic variation in the species and ecosystems and to ensure that selected areas are spread across the region to account for potential risks of catastrophe to areas in any one subregion. We attempted to meet goals for all applicable species and ecosystems within each subregion.

Targets were not identified in marine waters deeper than 40 meters, because very little data was available in these deeper waters. Goals ranged from 20-30% of existing distributions of the targets with most goals set at 30% (Figure 5.5.3a). The planning units were 750-hectare hexagon units covering terrestrial and marine environments. There were 8,107 planning units across the region.

In the Puget Trough ecoregion we compared unintegrated and integrated approaches for terrestrial and marine targets. Freshwater targets were included in the Puget Trough plan, but they were examined entirely separately from terrestrial and marine targets using different planning units, i.e., watersheds instead of hexagons. It is important to integrate freshwater with terrestrial and marine, but it was not possible to compare their integration analytically.

In the integrated analysis all nearshore marine and terrestrial targets were examined together in one analysis; the other analysis separated nearshore marine and terrestrial targets but retained the same exact planning units and the results were combined together (i.e., the 8107 hexagons of 750-hectares). We then compared the results of the analyses to determine if they were likely to affect ecological accuracy or economic efficiency (see Comparisons above).

Cook Inlet Basin Ecoregion, Alaska

In the Cook Inlet Basin Ecoregional Assessment (TNC, 2003) we examined integrated and unintegrated planning approaches across terrestrial, freshwater and marine environments. In the integrated analysis all terrestrial, freshwater and marine targets were examined together in one analysis; in the unintegrated analysis terrestrial, freshwater and marine environments were analyzed separately but retained the same exact planning units and the results were combined across the three environments.

The Cook Inlet Basin ecoregion is comprised of the low-lying basin surrounding Cook Inlet from the south side of the Alaska Range to Kachemak and Tuxedni Bays (Figure 5.5.4). It is bound on the east by the Kenai, Chugach and Talkeetna Mountains and on the west by the Alaska and northern Aleutian mountain ranges. The ecoregion includes the western half of the Kenai Peninsula, the Anchorage bowl, the western Cook Inlet lowlands, and the Susitna lowlands. The size of the ecoregion is 3,792,310 ha. Of this, the marine component is 886,200 ha and the terrestrial and freshwater portions are 2,906,110 ha. The lowlands of the ecoregion contain numerous lakes, estuaries and large river basins, including the drainages of the Kenai and Susitna rivers. These large rivers terminate in broad estuarine areas in the Cook Inlet. The Susitna River provides the greatest amount of freshwater input into Cook Inlet.

The Cook Inlet Basin Ecoregional Assessment was developed between 2000 and 2003, by The Nature Conservancy of Alaska and over 50 scientists and external partners. (TNC AK, 2003) It was developed with support from the U.S. Department of Defense/Fort Richardson and the U.S. Fish and Wildlife Service.

There were 7 different types of species and 245 different types ecosystems identified in the Cook Inlet ecoregion after accounting for potential stratification in the region. These were identified and mapped in this ecoregion across terrestrial, freshwater and marine environments (Figure 5.5.3b). A single set of contiguous planning units, 500 hectare hexagons, was developed across all three environments for a total of 7,516 planning units.

Representation goals were established for each target. The goals for most targets were 30% of their current distributions, with goals lowered for several widely occurring coastal systems to 20% of their current distributions. A suitability index was developed which reflected higher cost for representing targets in areas of likely lower ecological condition or site suitability. There were 24 different factors included in the suitability index including marine highways, wastewater permits, air runways, pollution spills, road density, dams, and oil platforms.

5.3 Results

Puget Trough Ecoregion

In the Puget Trough ecoregion, the integrated analysis was 4.7% more spatially efficient than the combination of planning units identified in the separate marine and terrestrial analyses (Figure 5.5.5). When the analyses were run separately a minimum of 2,962 exclusive planning units were identified to meet the objectives (Figure 5.5.5a). Some planning units that cross the shoreline were included in both terrestrial and marine analyses but any units selected in both analyses were only counted once. The minimum number of units identified among the best solutions in the integrated analysis was 2,830 planning units (2,122,500 ha) to meet objectives across both environments (Figure 5.5.5b). The integrated analyses met objectives with a total area that was 99,000 ha smaller than the areas identified to meet objectives from the combined, unintegrated analyses. Given the low variability in these different solutions (see Methods) even a 4.7%

change is significant. There are not readily discernible patterns in the planning units identified in the different analyses.

The integrated analyses, however, were less precise in meeting goals for all of the targets (Figure 5.5.6a). In the integrated analyses more goals were not met for biological targets, fewer goals for targets were adequately met and more targets vastly exceeded their goals than in the combination of the unintegrated analyses. This lack of precision was particularly evident for the marine targets; there was very little change in the precision of meeting goals for terrestrial targets between analyses.

Cook Inlet Basin Ecoregion

In the Cook Inlet ecoregion, the integrated analysis was 36.3% more spatially efficient than the combination of planning units identified in the separate terrestrial, freshwater, and marine analyses (Figure 5.5.7). When the analyses were run separately a minimum of 2,659 (1,329,500 ha) exclusive planning units were identified to meet the objectives (Figure 5.5.7a); these included 109 marine planning units, 1888 terrestrial planning units and 1083 freshwater planning units, minus 421 planning units that were selected in more than one environment. The minimum number of units identified among the best solutions in the integrated analysis was 1,951 planning units (975,500 ha) to meet objectives across all three environments (Figure 5.5.7b). The integrated analyses met objectives with a total area that was 354,000 ha smaller than the areas identified to meet objectives from the combined, unintegrated analyses.

There were clear differences in the pattern of areas identified between the unintegrated and integrated analyses. The planning units selected in the integrated analysis tended to cluster much more consistently and efficiently along riverine areas from the summit to the seas within the ecoregion as compared to the unintegrated analyses.

Unlike the Puget trough analyses, the integrated analyses were also more precise at meeting goals than the unintegrated analyses (Figure 5.5.6b). In the integrated analyses fewer goals were not met for biological targets, more goals for targets were adequately met and fewer targets exceeded their goals than in the combination of the unintegrated analyses.

5.4 Discussion

There is a need for better integration in conservation and management across environments. Conceptual approaches have been offered for how we can incorporate terrestrial, freshwater, and marine information systematic planning (e.g., Stoms 2005). Here we offer specific case examples of comparisons of different planning approaches that are informing ongoing conservation and management. It is rare to find quantitative comparisons of alternate planning approaches (Sala et al. 2002).

In the Puget Trough and Cook Inlet ecoregion, integrated planning across environments led to clear gains in spatial efficiency. These gains in spatial efficiency are likely to result in real gains in economic efficiency (i.e., reduced costs in conservation). If we can focus resources on fewer places and still meet all conservation objectives that is clearly preferable. Indeed even the costs of planning were reduced in these efforts, because just one suitability index was developed instead of three separate ones. The gains in spatial efficiency were not great for the Puget Trough ecoregion, although a difference of nearly 100,000 hectares is non-trivial. The gains in spatial efficiency were substantial in the Cook Inlet ecoregion example. The greater gains in efficiency in the Cook Inlet Basin ecoregion were attributable in part to the opportunity seeking efficiencies in integration across three environments not just two environments as in the Puget Trough ecoregion.

A potential benefit of integration in planning and management is a greater opportunity for ecological accuracy in accounting for the many connections in species, systems, and processes that span environments. True ecological accuracy is impossible to measure in a planning effort and as a proxy we used precision in meeting set goals for the biological targets. The results here were mixed. Goals were met a little less precisely for the integrated analyses in the Puget Trough, and they were met substantially more precisely for the integrated analyses in the Cook Inlet. In some respects, it is probably easier to meet goals more efficiently in unintegrated analyses, because those goals are set for biological targets that specifically occur in each environment. Having to meet goals across environments is likely an added constraint. This lack of constraint may explain the lower precision in integrated analyses in the Puget Trough. However given the huge loss in spatial efficiency in the unintegrated analyses in the Cook Inlet Basin ecoregion, it is not that surprising that many goals were vastly exceeded.

Most of the significant changes and advantages in integration in the Cook Inlet example occurred because of greater efficiency and accuracy in identifying terrestrial and freshwater sites. Overall the marine environment did not figure prominently in these analyses. As compared to freshwater and terrestrial environments, there were far fewer marine targets and much less overall spatial information on marine species, ecosystems and threats in the Cook Inlet ecoregion.

Constraints in Integration

There are real costs and limitations to developing integrated efforts as opposed to efforts that focus only on individual environments. The physical organization of science, conservation and management is often balkanized in separate departments, agencies, and organizations for the different environments. Integrated efforts will require greater coordination and communication among groups, departments, and agencies in planning and action. Integrated efforts may even require higher up front costs, because funds needs to be expended for planning and action in all environments.

More importantly in efforts that are fully integrated it is possible that the overall best focus of efforts will not always be those that will yield the greatest returns for any one environment. For example, the single best terrestrial sites (e.g., highest diversity) may not rank as the highest priority for conservation and management efforts in favor of sites that could yield greater combined benefits to all environments. This does not mean that those sites should not be conserved or managed, just that the priority level of that work could change. In subsequent reviews of the Puget Trough analyses by regional scientists and managers, the results of the unintegrated analyses tended to conform better to their views of the higher quality sites for conservation in the region, in particular for marine systems where there was less precision in meeting goals. It is possible that the integrated analyses are constrained to identify units that better meet goals across environments not just the best within any one environment.

Data availability has strong effects on site selection. There will likely always be less information available in the marine environment than in terrestrial or freshwater environments and this places real limits on how well efforts can be integrated.

Advantages and Practical Use of Analyses across Environments

The analyses in both regions were helpful and used by the planning teams and other scientists in developing the regional plans, but they were only an interim step. In both regions, there were clear advantages of the fully integrated analysis; it was more spatially efficient, nearshore marine and terrestrial targets were considered together in the land-sea interface, it was easier to manage target and site selection data within the database, only one suitability index was required, and in some cases precision in meeting goals was greater.

In the Puget Trough ecoregion, the planners ultimately used a partly integrated approach and there is practical advice from this approach. Both integrated and unintegrated analyses were consulted in the Puget Trough plan. In the end separate terrestrial and marine sites were selected by different groups of experts and these results were integrated post hoc by a joint committee that worked in particular to integrate sites in the coastal zone. The planners used the sites from the analyses coupled with oblique photos of the coast and expert input to align priority areas across the land sea boundary. This approach is less quantitative than just accepting Spexan model outputs, but planners, scientists, and stakeholders involved in the process clearly felt that this partly integrated approach was more ecologically accurate. Areas where terrestrial environmental processes supported marine conservation objectives were highlighted, such as places where coastal sandy "feeder bluffs" provided nutrients and sediments to adjacent marine conservation targets (e.g., eelgrass). There were other areas where integration was enhanced simply by better aligning the priority sites selected from the unintegrated Spexan analyses. This alignment sometimes involved just changing the shapes of particular sites to better connect them. In some cases, sites were moved entirely to accommodate better connections among priority sites. These changes were possible for sites that primarily contained common targets (e.g., salt marshes) and where it was possible to meet representation goals for these common targets at many potential priority sites.

There will often be significant barriers to conducting fully integrated planning exercises such as institutional barriers, information gaps, or lack of available expertise. Under circumstances such as these, plans for individual environments can still be informed by information from other environments (partially integrated). For example, a preference can be given for selecting priority areas in one environment that are adjacent to the priority areas (e.g., existing reserves or parks) in another environment. An existing plan may have identified a set of priority freshwater sites specifically designed to efficiently represent freshwater aquatic systems and species. Those results can be used to adjust analyses of marine priorities giving a preference to capturing marine targets in areas where the freshwater inputs are already receiving conservation attention. It is also possible to favor locating terrestrial conservation efforts in areas adjacent to sites of marine biodiversity significance or vice versa.

Planning efforts can also be partly integrated by setting goals for particular species or systems in one environment not just because they are important in their own right, but because their presence is connected and vital to other targets across environments. For example, a goal of preserving say 10,000 horseshoe crabs in a Delaware Bay plan may be enough to ensure their population viability. However, a much higher goal for crabs may be necessary to ensure that sufficient numbers of female crabs come ashore and lay the eggs that have been shown to be vital to sustaining shorebirds traversing the Atlantic flyway.

One method that may help to improve precision and accuracy in analyses with a single set of contiguous planning units is to split planning units at the shoreline. Splitting the analysis units at the coast creates a clear boundary and allows users to make independent decisions for capturing specifically marine or specifically terrestrial targets. This independence also reduces biases that might occur for the selection of just those particular units that overlap environments, because they contain terrestrial, freshwater and marine targets in the same unit. This technique can be extended to account for the fact that certain areas have significant linkages across the land-sea interface and we can explicitly manipulate how much emphasis to place on selecting adjacent cells across the land sea boundary.

Conclusions

The examples illustrate that integrated planning is possible and most importantly that there are gains in spatial efficiency, occasional improvements in precision, and some gains in overall planning efficiency (i.e., the time required for the planning team to complete analyses across environments). There are ecological and economic advantages to combining marine, terrestrial, and freshwater site selection into a single analysis or decision support framework. While a fully integrated analysis may not be appropriate in all cases, analyses which are done separately across terrestrial, freshwater and marine environments can benefit from being informed by analyses in adjacent environments and there are practical ways to achieve better integration.

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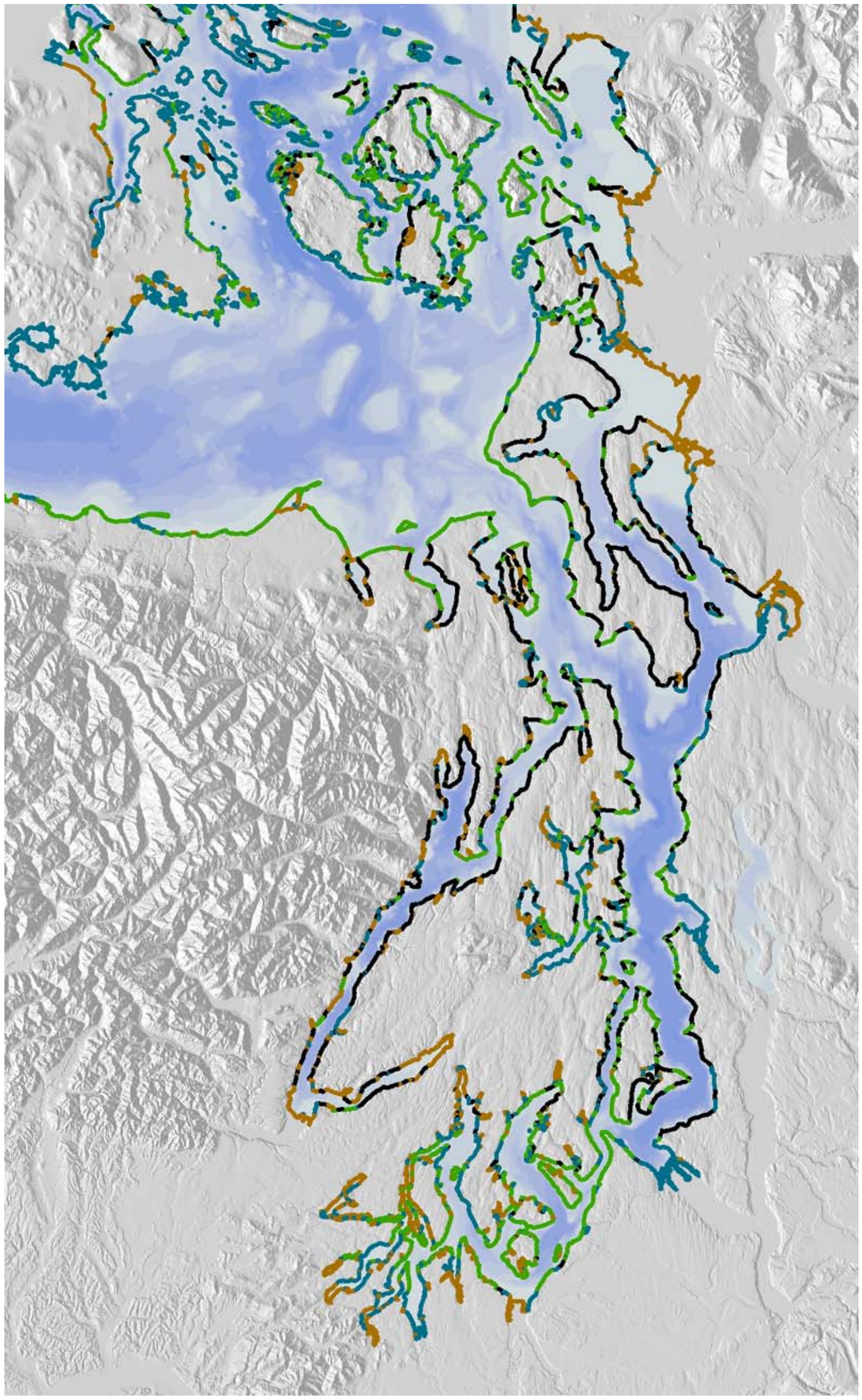
5.5 Figures

5.5.1 Basic approach to systematic planning

Action	Description
Identify objectives	e.g., to represent and conserve a full range of a region's biodiversity for conservation
Select targets	The focus of the planning effort, e.g., species and ecosystems
Identify goals	The amount (e.g., abundance, area) of the targets required to meet objectives
Identify suitability factors	Factors that can affect the (i) cost or (ii) suitability of an area for meeting objectives or the (iii) viability and (iv) threats to the targets (e.g., human population density, shipping lanes)
Develop spatial database	Spatially-explicit information on the targets and suitability factors
Establish analysis units	The targets and suitability factors are tracked within these units. The units may be natural (e.g., bays) or artificial (e.g., a grid of equal-sized units)
Select sites	To achieve the stated goals and objectives. Site-selection tools (e.g., Marxan) are commonly used at this stage.
Develop strategies	To meet conservation and management objectives at selected sites
Review results	Peer review and revision with experts and stakeholders

5.5.2 Example of the distribution of major ecosystems in the Puget Trough ecoregion. Saltmarshes are in brown, kelps and seagrasses are in green, seagrasses alone are in black, and other shoreline ecosystem types are in blue.



5.5.3 Targets in a) the Puget Trough and b) Cook Inlet ecoregions

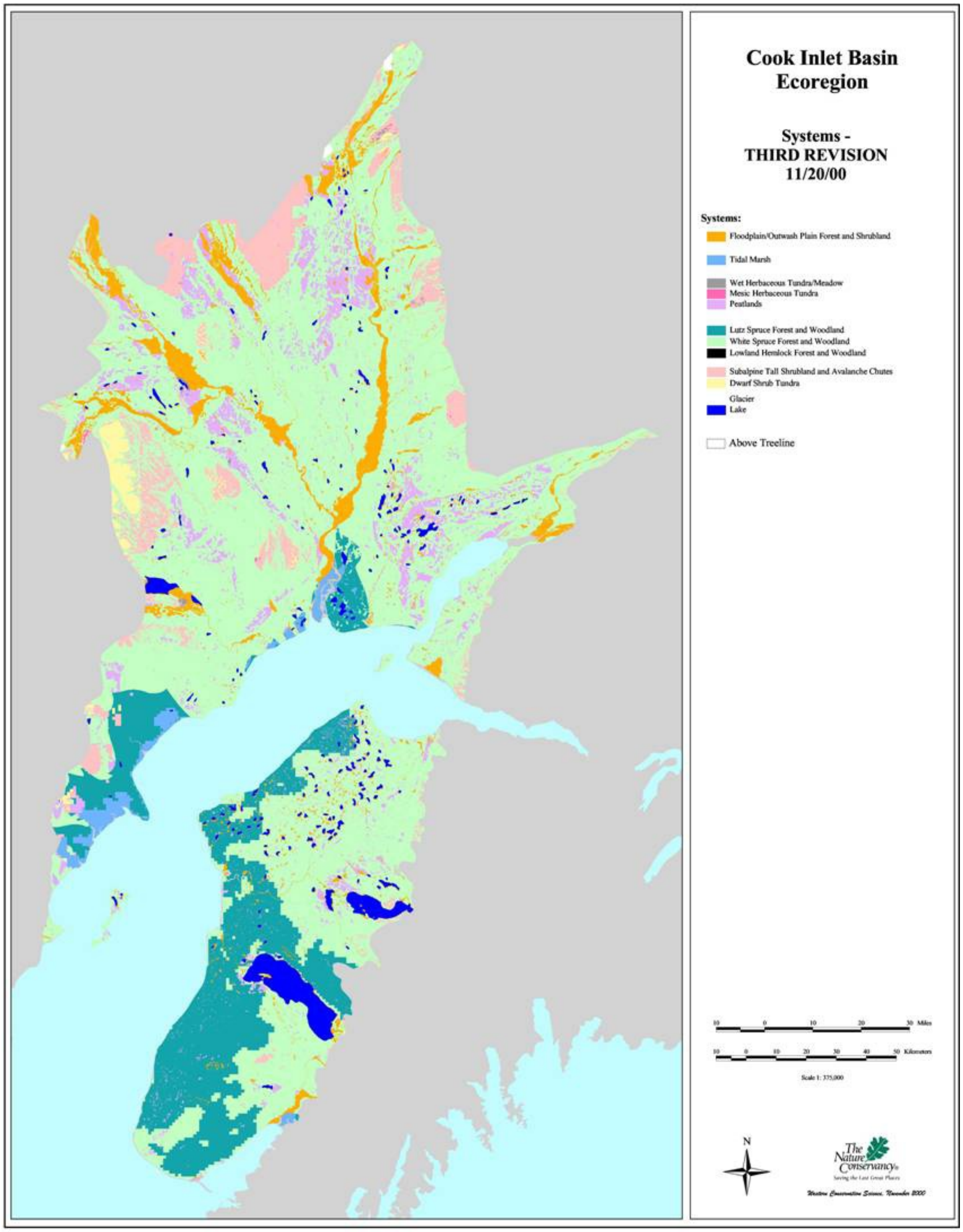
a. Puget Trough Ecoregion

Type of Target	# of targets	Goals	Examples
Terrestrial ecological systems	19	30%	
Vascular plants	239	30 - 100%	
Non-vascular plants	56	30 - 100%	
Terrestrial animals	127	30 - 100%	20 mammals, 45 birds, 8 reptiles, 16 amphibians, 31 insects, 6 molluscs, 1 earthworm
Nearshore marine ecological systems	40	30%	
Marine animals	68	20 - 60%	11 fish, 8 marine mammals, 12 seabirds, 26 invertebrates, 11 plants and algae
TOTAL	549		needs to be verified with terrestrial planning team

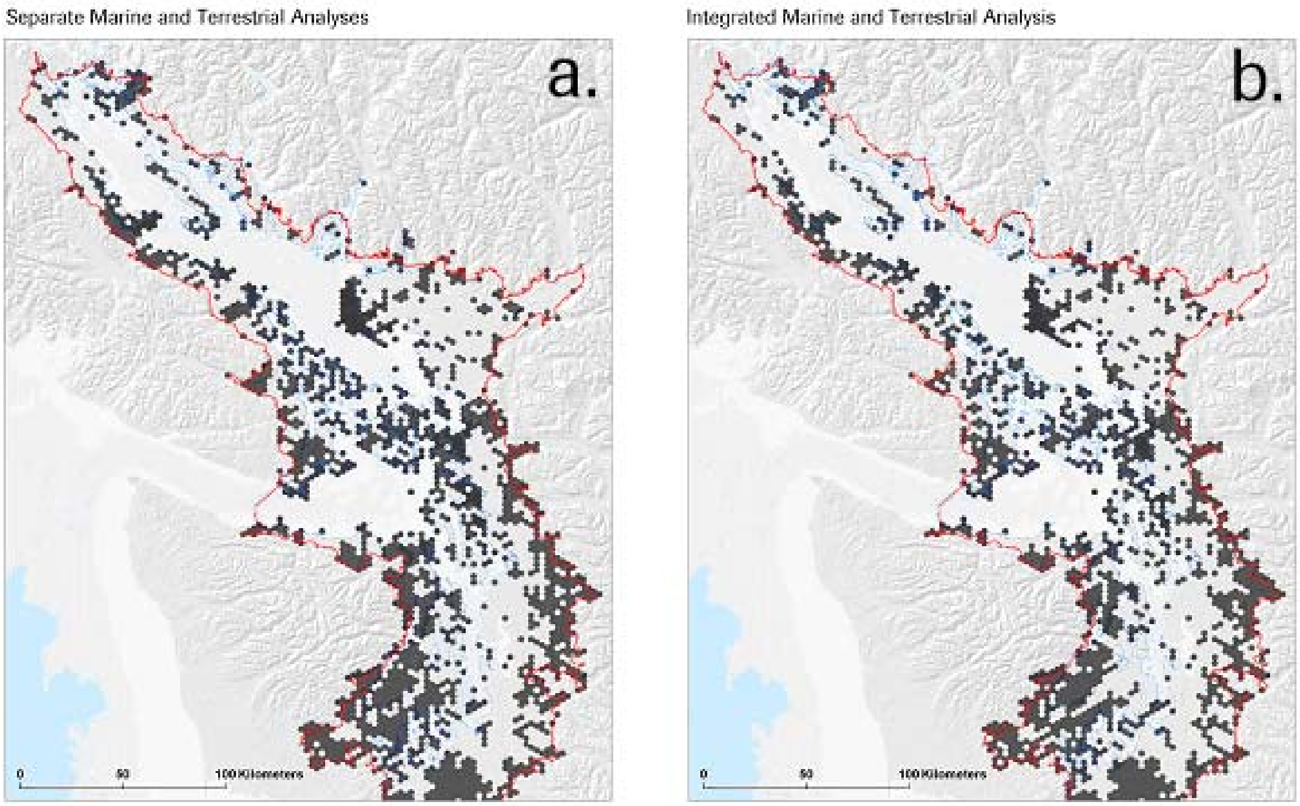
b. Cook Inlet Ecoregion

Type of Target	# of targets	Goals	Example of targets
Ecological systems	245		
Terrestrial systems	10	30%	Below Tree-line Fluvian Rolling Plain, Wet
Ecological Land Types	174	30%	Lutz Spruce Forest and Woodland
Freshwater Aquatic systems	51	30%	Glacial mainstem river, stream on moraine
Coastal marine systems	10	20-30%	Kelp forest, tidal marsh, mussel beds, estuaries
Species	6	6	
Birds	1	100%	*Aleutian Tern, Aegolius funereus*, *Polysticta stelleri*, *Chen caerulescens*
Marine mammals	2	100%	Harbor Seal, Beluga Whale
Plants	3	100%	Puccinella triflora
Species aggregations	1	100%	Shorebird aggregation
TOTAL	252		

5.5.4 Example of the distribution of major ecosystems in the Cook Inlet ecoregion



5.5.5 Results of the Puget Trough ecoregional analyses: a) unintegrated and b) integrated



5.5.6 Precision in meeting goals for targets. Comparison between analyses of the number and percent of biological targets that did not meet goals and were underrepresented (<97% of goal met); targets that met goals and were adequately represented (97-130% of the goal was met); targets that vastly exceeded goals and were over-represented (>130% of goal met).

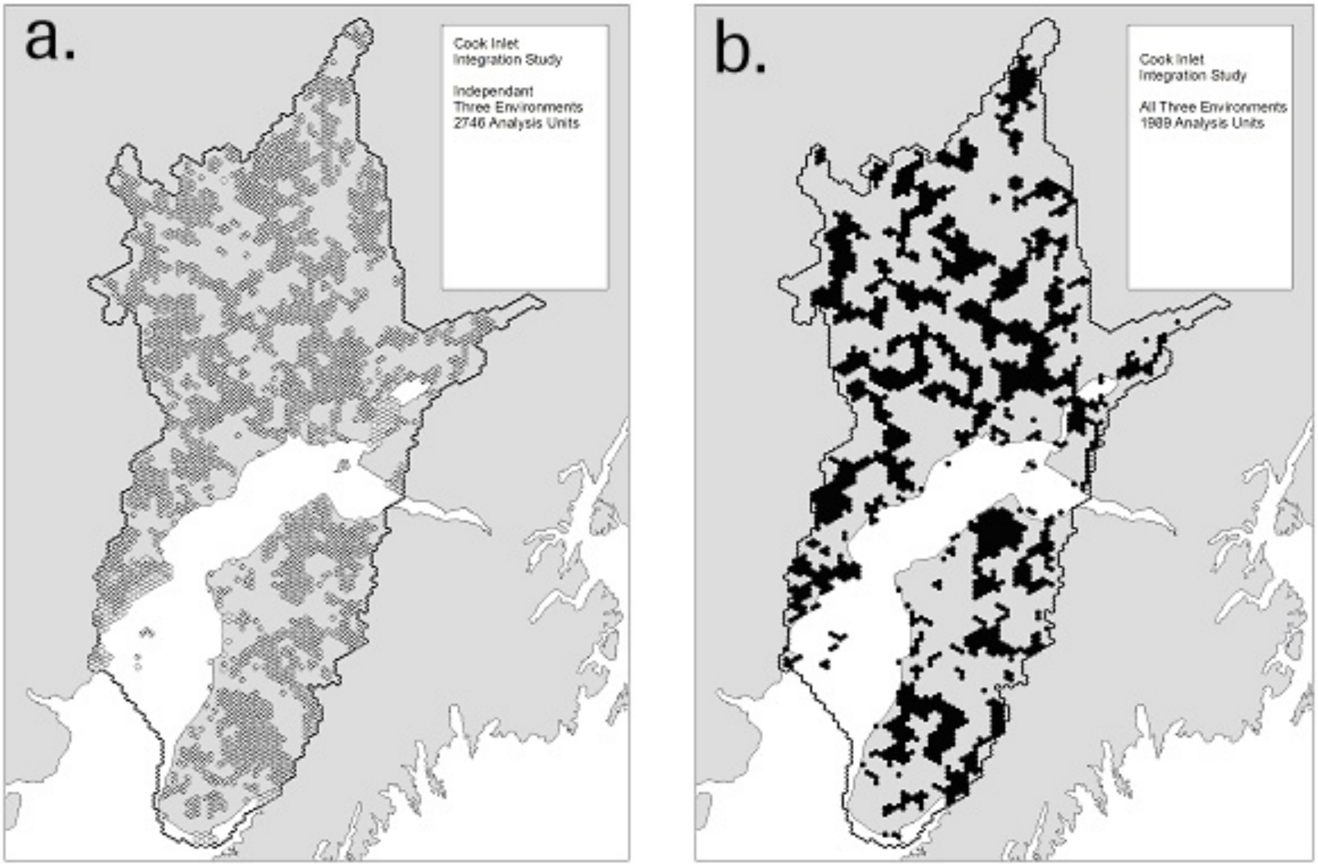
a. Puget Trough Ecoregion

Analysis	Goals Not Met	Goals Met	Goals Exceeded
Integrated	23 (4%)	465 (78%)	105 (18%)
Unintegrated	32 (5%)	496 (84%)	65 (11%)
marine	6 (5%)	85 (71%)	28 (24%)
terrestrial	26 (5%)	411 (87%)	37 (8%)

b. Cook Inlet ecoregion

Analysis	Goals Not Met	Goals Met	Goals Exceeded
Integrated	12 (2%)	289 (53%)	247 (45%)
Unintegrated	24 (4%)	129 (24%)	395 (72%)
marine	0	19 (73%)	7 (27%)
terrestrial	23 (5%)	107 (23%)	341 (72%)
freshwater	1 (2%)	3 (6%)	47 (92%)

5.5.7 Results of Cook Inlet ecoregional analyses: a) unintegrated and b) integrated



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Appendix I: Use of the Geodatabase

The ecoregional planning process produces large amounts of information that must be organized for utilization by The Nature Conservancy and its partners. There are many ways to compile and represent these data. We have been testing the Environmental Systems Research Institute's (ESRI) database format called the geodatabase. The geodatabase is capable of organizing large amounts of tabular and spatial information. The tabular data are formatted to database standards and are available through Microsoft Access (version 2000 or higher) or ArcGIS (version 8.3 or higher) software. The spatial data are also organized in database format and are available through ArcGIS. The personal geodatabase, which is capable of storing upwards of two gigabytes of information, allows the user to take advantage of both software programs in designing a database for conservation planning purposes. We have utilized the Relational Database Management System (RDBMS) of the geodatabase data model to explicitly link tabular data and targets, goals, geography, and MARXAN input files with the spatial data representing them in one repository. This initial version of the Pacific Northwest Coast geodatabase focuses on data used for the preliminary nearshore conservation portfolio, but we have also spawned additional geodatabases for our land/sea threats and integration work.

We consider the geodatabase and its functionality to be a key component of our decision support system (DSS) for conservation planning at multiple scales. DSS covers a wide variety of systems, tools and technologies (Power 1997). In the context of conservation planning we define DSS as “a toolbox for planners and managers that can be readily used when making natural resource decisions”. The DSS is a useful and inclusive term for many types of information systems and includes a wide variety of tools depending on the issues being addressed. The context introduced here includes the databases and tools needed to develop spatial analyses in order to assist in marine conservation planning. We recognize three aspects of this DSS: 1) spatial database development, 2) optimization tools and simulation models, and 3) training. Here we address the first and second aspects of the DSS.

Perhaps the fundamental concept of the geodatabase is its uniform repository of geographic data, or the ability to store many types of geographic data. Since all related data are managed in one database, updates and distribution of ecoregional information product can be made much easier. Developing standards within this centralized spatial database is important when considering its ease-of-use as well as relating it to adjacent ecoregional or other planning efforts. We have an initial naming convention for files in the geodatabase, and have designed the database around relationships between assessment units and target information. We also have plans to standardize attribute names to further advance our database design. Below represents a list of the spatial contents within the geodatabase:

- Boundaries ("bnd" prefix): Pacific Northwest Coast ecoregion, marine region

- Assessment units ("au" prefix): nested grids (400-hectare nearshore, 1,600-hectare offshore), estuary polygons and HUC watersheds

- Ecoregion-wide targets ("nwc" prefix): original estuary, shoreline and herring spawn data sets before intersecting the data with the assessment units

- Conservation targets ("tgt" prefix): estuary and shoreline coarse filter data intersected with nested grid assessment units as well as nearshore marine species including forage fish, marine mammals, seabirds, and intertidal invertebrates

- Suitability or cost index ("cst" prefix): individual and compiled data sets on cost factors including onsite shoreline impacts, adjacent upland impacts and management designations as well as adjacent nearshore impacts and management designations

- MARXAN input files and target lists ("tbl" prefix): files that represent the amount of shared boundary between assessment units, multiple cost factors built into a suitability index, ecosystem, habitat, and species target distributions by assessment unit, and a target/goal table

- Portfolio solutions ("prt" prefix): initial nearshore site selection ("best" and "sum of summed" solutions) and portfolio design (incorporated expert review and initial integration) before land/sea integration

In addition we have created subsequent geodatabases from the original Pacific Northwest Coast repository for specific threats and integrated analyses:

- Threats information ("trt" prefix): tabular and spatial information pertaining to ongoing and future threat factors. This will build off of the existing suitability index that depicts current conditions, and will include information for subsequent analyses at multiple scales.

In all geodatabase work we also attempt to standardize names for queries in Access:

- Access append queries ("qry" prefix): queries that update the site selection input files as new information becomes available and older data is refined

- Access append summary queries ("sum" suffix): after completing a planning exercise there is a need to go back through the target data,summarize distributions and calculate how well each one did in accomplishing their conservation goals. As many targets are stratified by ecoregional subsections we conduct these summaries based on un-stratified targets.

Use of the geodatabase has also allowed us to design a spatial database that is linked to the optimization tool MARXAN for evaluating alternatives or multiple scenarios in selecting high priority conservation areas. By dynamically linking the geodatabase to site-selection algorithms, planners can easily and efficiently explore the impacts of potential conservation actions that affect target status and assess how priorities may shift spatially given a sequence of actions (Merrifield et al, in prep). For example, if you knew that you would have funding to complete ten land acquisition projects and engage in a county general plan that will confer additional land protection in the next year, you could quantify the impact of these actions on the conservation status of your targets and re-run the site-selection algorithm to help re-orient your priorities. This "re-shuffling the deck" functionality would help ensure that organizations stay dynamic, adaptive and directed toward stated conservation goals. This concept can be used in the converse situation as well. For instance, if a part of the region has been identified for development or unsustainable exploitation, planners can examine how that re-allocates priorities.

A regional geodatabase can function to inform multiple scales of planning. As a primary objective of ecoregional planning, the identification of high priority conservation areas needs to be verified through ground-truthing and finer resolution planning efforts. Regional spatial information may still have utility at finer scales of analysis, especially in the marine environment where there is a paucity of data at all scales. The addition of new information can either be connected to or spawned from the central geodatabase used in the original planning effort.

Using a geodatabase has allowed us to archive spatial data, maintain relationships between conservation targets, and distribute information to partners. Further research into geodatabase functionality includes the ability to identify relationships between targets, suitability factors, and both nearshore and watershed assessment units used for analysis. The geodatabase attempts to link features of a physical data model closer to its logical data model (Zeiler 1999). In other words, the nature of the geodatabase connects the physical structure of features with their specific qualities and relationships between them. For example, relationships can be established between watershed characteristics and associated coastal/nearshore conditions. The relationship of a coastal watershed to its assemblage of shoreline ecosystems can be explicitly linked giving us the opportunity to associate watershed (e.g., size and flow accumulation) with nearshore parameters (e.g., wave energy and salinity).

We are also exploring the use of the geodatabase for rolling-up or connecting multiple ecoregional assessments. Although the personal geodatabase is limited to two gigabytes of information, an enterprise geodatabase that is stored on a central server has more storage capacity as well as version control from multiple users. By their design, enterprise geodatabases can accommodate very large sets of features without tiles or other spatial partitions. With version control functionality multiple users can edit the geographic data simultaneously. The geodatabase data model may be implemented within a Relational Database Management System (RDBMS), supporting work flows where multiple people can simultaneously edit features and reconcile any conflicts that emerge.

Many ecoregional planning teams recognize the power of assembling ecological and human impact data in evaluating a region's biodiversity. This collection of information is vital to our understanding of the distribution of ecosystems, habitats, species, impacts and threats to the environment as well as identifying a suite of important places to focus conservation attention. This information fuels the decision making process, and needs to be flexible and updateable to account for the changing landscape.

As an organizational tool alone, the geodatabase functions to help planning teams assemble the information into a distinct product, making it easily accessible and able to distribute across the Conservancy and to partners. Until recently, planning teams have stored ecoregional data in files which sit unused once a final conservation portfolio is assembled. This fails to utilize the power of the information collected, although it is these data that often interest partners most. In many cases the final conservation portfolio serves the purpose of The Nature Conservancy alone in conveying a vision for conservation, but often partners that participated in the process cannot endorse its product without the underlying data. For instance, the ecoregional portfolio often crosses more than one state or federal jurisdiction, or international boundary, while a partner agency must focus on the portion of the ecoregion that they manage. In such cases, sub-regional analyses within the ecoregion may be an important next step in conservation planning with partners. Any ecoregional data model needs to incorporate this kind of flexibility where the underlying information can be easily utilized while its relationships are maintained.