CHAPTER 17

Ecosystem service assessments for marine conservation

Anne D. Guerry, Mark L. Plummer, Mary H. Ruckelshaus, and Chris J. Harvey

17.1 Introduction

Humans always have benefited from marine ecosystems—either obviously in the form of food resources, or more subtly in the form of cultural and recreational opportunities. For example, 80–85 million tons of fish were landed in marine capture fisheries worldwide in 2006, and fish account for approximately 15% of the annual animal protein consumption by humans (FAO Fisheries Department 2009). A growing recognition of the degradation of global marine ecosystems has led to numerous calls for a shift toward more holistic, ecosystem-based management of marine systems (Pew Oceans Commission 2003; US Commission on Ocean Policy 2004; Council on Environmental Quality 2009). Ecosystem-based management is a coordinated effort to manage the diverse human impacts that affect an ecosystem to ensure the sustainability of the ecosystem services it provides (Rosenberg and Mcleod 2005). Two key aspects of ecosystem-based management are relevant here. First, ecosystem-based management fundamentally recognizes the inseparability of human and ecological systems. Human well-being is derived from ecosystems through ecosystem services and, in turn, human behavior affects natural systems. Second, ecosystem-based management is inherently multifaceted, encompassing suites of services, rather than the traditional approach of sector-by-sector management. Importantly, the framework of ecosystem services can provide performance metrics for different management strategies that attempt to balance multiple objectives by allowing for the explicit examination of trade-offs in ecosystem services provided under alternative management scenarios (National Research Council 2004).

Using an ecosystem services framework also has the potential to draw a larger and more diverse population of people to marine and other conservation efforts, beyond those who value the environment purely for its direct uses. For example, many residents are drawn to the Puget Sound region, USA because of the sound’s physical beauty and concomitant aesthetic benefits to their well-being. Indeed, existence values have been found to be among the “most important” benefits provided by the Puget Sound system (Iceland et al. 2008). If such cultural values can be included in tallies of the consequences of ecosystem protection, conservation efforts are likely to engage a greater fraction of the population. Helping people to see the many ways their well-being is affected by marine and coastal environments is key to the success of conservation.

In principle, marine ecosystem services are not fundamentally different from their terrestrial counterparts. In practice, however, the valuation and mapping of ecosystem services in marine environments is not as well developed as it is for terrestrial ecosystems. As described in Chapters 4–13, there have been some early successes applying ecosystem service mapping and modeling tools in diverse terrestrial and freshwater settings. These approaches and models all start with basic land cover and land-use data layers. The same approach can work for marine environments—marine systems have patchy habitats that provide flows of ecosystem services, and management actions can alter those habitats.
and flows. Several challenges must be addressed, however. Maps of habitat type and habitat use are much harder to come by in marine systems than they are on land. We cannot readily “see” many parts of the marine ecosystem and its habitat types using satellite imagery or other remote sensing technology. Moreover, marine habitats and the processes that maintain them are more transient and three-dimensional than their terrestrial counterparts, and associations between particular species and habitats are harder to document.

Another challenge stems from the ways in which humans interact with marine environments. While fishery harvest, one of the most important marine ecosystem services, is straightforward to measure, its ecological effects and potential impacts on other ecosystem services are harder to discern. In addition, many of our actions that affect the marine environment take place on land. For example, coastal development; land-use practices that produce nutrient, sediment, and pathogen inputs to freshwater; and increases in impervious surfaces can severely degrade nearshore marine systems (Carpenter et al. 1998; Mallin et al. 2000; Diaz and Rosenberg 2008). Incorporating an ecosystem service perspective into marine management, then, facilitates integration of terrestrial and marine policies, which have been historically disconnected.

Fortunately, there are advanced aspects of marine science that will provide a good foundation for ecosystem service analyses. In particular, although basic mapping data are less refined in marine environments, marine science has a rich ecosystem-based modeling tradition to draw on for quantifying ecosystem services. For example, modeling for fisheries management (e.g., Christensen and Walters 2004; Pauly et al. 2000; Fulton et al. 2004a, b) and water use impacts in the Everglades and Florida Bay (e.g., US Geological Survey 1997) provide sophisticated system and food web models that can be extended to evaluate a more comprehensive suite of human activities and ecosystem services.

### 17.2 Ecosystem services provided by marine environments

Global oceans provide a wealth of ecosystem benefits that span all four major categories of services identified by the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment 2005): provisioning, regulating, cultural, and supporting services (Table 17.1). Marine ecosystems provide goods and

<table>
<thead>
<tr>
<th>Subcategory</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Provisioning services</strong></td>
<td></td>
</tr>
<tr>
<td>Food</td>
<td>Capture Fisheries</td>
</tr>
<tr>
<td></td>
<td>Aquaculture</td>
</tr>
<tr>
<td></td>
<td>Wild foods</td>
</tr>
<tr>
<td>Fiber</td>
<td>Mangrove wood (construction, boat-building), seagrass fiber</td>
</tr>
<tr>
<td>Biomass fuel</td>
<td>Mangrove wood (charcoal), biofuel from algae</td>
</tr>
<tr>
<td>Water</td>
<td>Individual salmon stocks, marine diversity for bioprospecting</td>
</tr>
<tr>
<td>Genetic resources</td>
<td>Medicines</td>
</tr>
<tr>
<td>Biochemicals, natural medicines, and pharmaceuticals</td>
<td>Food additives</td>
</tr>
<tr>
<td><strong>Regulating services</strong></td>
<td></td>
</tr>
<tr>
<td>Air quality regulation</td>
<td>Sea salt and spray help cleanse the atmosphere of air pollution*</td>
</tr>
<tr>
<td>Climate regulation</td>
<td>Major role in global CO₂ cycle</td>
</tr>
<tr>
<td>Water regulation</td>
<td>Natural stormwater management by coastal wetlands and floodplains</td>
</tr>
<tr>
<td>Erosion regulation</td>
<td>Nearshore vegetation stabilizes shorelines</td>
</tr>
</tbody>
</table>

(continues)
services from both biotic (e.g., depend on food webs) and abiotic (e.g., depend on the presence of seawater) aspects of natural capital. Assessment reports within (Agardy et al. 2005) and based on the Millennium Ecosystem Assessment (UNEP 2006) and other synthesis documents (Peterson and Lubchenco 1997; Costanza 2000; Patterson and Glavovic 2008; Wilson and Liu 2008) provide useful overviews of these services, as do descriptions of the particular services provided by fish populations (Holmlund and Hammer 1999), coral reef ecosystems (Moberg and Folke 1999), and mangroves (Ronnback 1999).

Provisioning services include the most high-profile marine ecosystem services such as food from capture fisheries, aquaculture, and wild foods. On average, each person alive in 2006 ate 16.7 kg of fish that year (18% of that total came from marine aquaculture; the proportion from capture fisheries is difficult to calculate given non-food uses of wild fish) (FAO Fisheries Department 2009). Some of the less obvious provisioning services include timber and fiber from mangroves and seagrass beds, and biochemicals for cosmetics and food additives. The potential also exists for developing novel natural products from marine species with medical applications (Carté 1996). In addition, the ocean may become an important energy source: biofuels from algae and power generation from wave and tidal energy have potential for more widespread use. And finally, the world’s oceans provide the highways for the global shipping trade.

Marine systems also are responsible for a wide range of regulating services. Most prominent of these is natural hazard regulation. As was vividly highlighted by the human losses wrought by the 2004 Asian tsunami and 2005 hurricanes on the Gulf Coast of the USA, coastal and estuarine wetlands have value for their ability to reduce storm surge elevations and wave heights (Danielsen et al. 2005; Travis 2005; Box 17.1). Other regulating services

<table>
<thead>
<tr>
<th>Subcategory</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water purification and waste treatment</td>
<td>Uptake of nutrients from sewage wastewater, detoxification of PAH’s by marine microbes, sequestration of heavy metals</td>
</tr>
<tr>
<td>Disease regulation</td>
<td>Natural processes may keep harmful algal blooms and waterborne pathogens in check</td>
</tr>
<tr>
<td>Pest regulation</td>
<td>Grazing fish help keep algae from overgrowing coral reefs</td>
</tr>
<tr>
<td>Pollination/assistance of external fertilization</td>
<td>Innumerable marine species require seawater to deliver sperm to egg</td>
</tr>
<tr>
<td>Natural hazard regulation</td>
<td>Coastal and estuarine wetlands and coral reefs protect coastlines from storms</td>
</tr>
<tr>
<td>Cultural services</td>
<td></td>
</tr>
<tr>
<td>Ethical values</td>
<td>Non-use</td>
</tr>
<tr>
<td>Existence values</td>
<td>Non-use</td>
</tr>
<tr>
<td>Recreation and ecotourism</td>
<td>Non-consumptive use</td>
</tr>
<tr>
<td>Supporting services</td>
<td></td>
</tr>
<tr>
<td>Nutrient cycling</td>
<td>Major role in carbon, nitrogen, oxygen, phosphorus, and sulfur cycles</td>
</tr>
<tr>
<td>Soil formation</td>
<td>Many salt marsh surfaces vertically accrete; eelgrass slows water and traps sediment</td>
</tr>
<tr>
<td>Primary production</td>
<td>~40% of global NPP**</td>
</tr>
<tr>
<td>Water cycling</td>
<td>96.5% of earth’s water is in oceans***</td>
</tr>
</tbody>
</table>

* Rosenfeld et al. (2002).
*** Melillo et al. (1993).
†† The taxonomy of services is adapted from the Millennium Ecosystem Assessment (2005).
Box 17.1 Nonlinear wave attenuation and the economic value of mangrove land-use choices

Edward B. Barbier

Although most ecologists have concluded that ecosystem size and functional relationships are non-linear, the lack of data or mapping of these relationships has often precluded estimating how the value of an ecosystem service varies across an ecological landscape. However, recent collaborations between ecologists, hydrologists and economists have demonstrated this effect for the wave attenuation function of mangroves, which in turn impacts on the land-use choices for conserving or developing mangrove forests.

Barbier (2007) conducted a comparison of land-use values between various mangrove ecosystem benefits and conversion of the mangrove to shrimp ponds in Thailand. He found that all three ecosystem services - coastal protection, wood product collection and habitat support for off-shore fisheries—have a combined value ranging from $10,158 to $12,392 ha⁻¹ in net present value terms over the 1996 to 2004 period of analysis, and that the highest value of the mangrove by far is its storm protection service, which yields an annual benefit of $1,879 ha⁻¹ annually, or a net present value of $8,966 to $10,821.

But what if these per hectare values for mangroves were used to inform a land-use decision weighing conversion of an entire mangrove ecosystem to shrimp aquaculture? For example, deciding how much of a mangrove forest extending 1000 m seaward along a 10-km coastline to convert to shrimp aquaculture may depend critically on whether all the mangroves in the 10 km² ecosystem are equally beneficial in terms of coastal storm protection (Barbier et al. 2008).

Suppose that it is assumed initially that the annual per ha values for the various ecosystem benefits are “uniform,” and thus vary linearly, across the entire 10 km² mangrove landscape. Following this assumption, a mangrove area of 10 km² would have an annual storm protection value of 1000 times the $1,879 ha⁻¹ “point estimate,” which yields an annual total benefit estimate of nearly $1.9 million. Barbier et al. (2008) show how this assumption translates into a comparison of the net present value (10% discount rate and 20-year horizon) of shrimp farming to the three mangrove services - coastal protection, wood product collection and habitat support for off-shore fisheries—as a function of mangrove area (km²) for the example of a 10 km² coastal landscape. Figure 17.A.1 shows the comparison of benefits. The figure also aggregates all four values to test whether an “integrated” land-use option involving some conversion and some preservation yields the highest total value. When all values are linear, as shown in the figure, the outcome is

![Figure 17.A.1](image-url)  
Linear ecosystem service returns from mangroves.
Box 17.1 continued

a typical “all or none” scenario; either the aggregate values will favor complete conversion or they will favor preserving the entire habitat. Because the ecosystem service values are large and increase linearly with mangrove area the preservation option is preferred. The aggregate value of the mangrove system is at its highest ($18.98 million) when it is completely preserved, and any conversion to shrimp farming would lead to less aggregate value compared to full preservation, thus any land-use strategy that considers all the values of the ecosystem would favor mangrove preservation and no shrimp farm conversion.

However, not all mangroves along a coastline are equally effective in storm protection. It follows that the storm protection value is unlikely to be uniform across all mangroves. The reason is that the storm protection “service” provided by mangroves depends on their critical ecological function in terms of “attenuation” of storm waves. That is, the ecological damages arising from tropical storms come mostly from the large wave surges associated with these storms. Ecological and hydrological field studies suggest that mangroves are unlikely to stop storm waves that are greater than 6 m (Forbes and Broadhead 2007; Wolanski 2007; Alongi 2008; Cochard et al. 2008). On the other hand, where mangroves are effective as “natural barriers,” against storms that generate waves less than 6 m in height, the wave height of a storm decreases quadratically for each 100 m that a mangrove forest extends out to sea (Mazda et al. 1997; Barbier et al. 2008). In other words, wave attenuation is greatest for the first 100 m of mangroves but declines as more mangroves are added to the seaward edge.

Barbier et al (2008) employ the non-linear wave attenuation function for mangroves based on the field study by Mazda et al. (1997) to revise the estimate of storm protection service value for the Thailand case study. The result is depicted in Figure 17.A.2.

The storm protection service of mangroves still dominates all values, but small losses in mangroves will not cause the economic benefits of storm buffering by mangroves to fall precipitously. The consequence is that the aggregate value across all uses of the mangroves, shrimp farming and ecosystem values, is at its highest ($17.5 million) when up to 2 km² of mangroves are allowed to be converted to shrimp aquaculture and the remainder of the ecosystem is preserved.

Taking into account the “nonlinear” relationship between an ecological function and the value of the ecosystem service it provides can therefore have a significant impact on a land-use decision at the landscape scale. Other ecosystem services, including those for mangroves, are likely to have similar effects. For example, a study of the nursery habitat function of mangroves in the Gulf of California, Mexico reveals that the function’s influence on the productivity of off-shore fisheries does not scale-up in direct proportion to the area of the mangrove forests in the nearby lagoons (Aburto-Oropeza et al. 2008).

Figure 17.A.2 Nonlinear ecosystem service returns from mangroves.
provided by marine systems include the transformation, detoxification, and sequestration of wastes (Peterson and Lubchenco 1997).

Rich cultural services are provided by marine systems. Human coastal communities—both native and non-native—often define their identities in relation to the sea. In the U.S., people love to live near the ocean; one study predicts average increases of 3,600 people a day moving to coastal counties through 2015 (Culliton 1998). Globally, coastal tourism is a key component of many economies (Box 17.2). It is one of the fastest growing sectors of tourism, and is one of the world’s most profitable industries (United Nations Environment Programme 2006).

Finally, the oceans provide essential supporting services that underpin many of the world’s ecological functions. The oceans are the center of the global water cycle; they hold 96.5% of the earth’s water (Gleick 1996) and are a primary driver of the atmosphere’s temperature, moisture content, and stability (Colling 2001). Oceans are also key players in the global cycles of carbon, nitrogen, oxygen, phosphorus, sulfur, and other key elements (Peterson 1997).

---

**Box 17.2 Valuation of coral reefs in the Caribbean**

*Emily Cooper and Lauretta Burke*

In the Caribbean, nearly 70% of coral reefs are threatened by human activities—including over-fishing, dredging, sewage discharge, increased runoff from agricultural activities, and coastal development (Burke and Maidens 2004). Degradation of reefs not only results in a tremendous loss of biodiversity but also leads to a decline in the services they provide to coastal communities, resulting in lost revenue from declining tourism and fishing, increased poverty and malnutrition, and increased coastal erosion.

Many of these damaging activities occur because an individual or group seizes an immediate benefit, without knowing or caring about the long-term consequences. Quantifying the value of ecosystem services provided by reefs can help to facilitate more sensible, far-sighted decision-making by drawing attention to the economic benefits associated with reefs, and by demonstrating the true costs of poor coastal management. In 2005 the World Resources Institute (WRI) launched a project to assess the economic contribution of three reef-related ecosystem services to countries in the Caribbean: reef-related fisheries, tourism, and shoreline protection. These three services were chosen because they are (a) relatively easy to measure using published information, (b) easily understood by politicians and decision-makers, and (c) especially important to local economies. National-level studies have been completed for St. Lucia, Tobago, and Belize. In Tobago, one of two pilot sites, the project estimated the value of these three services at US$62–78 million per year (Burke et al. 2008).

---

**Reef-related tourism and fisheries**

Tourism is Tobago’s largest economic sector, contributing 46% of GDP and employing 60% of the workforce (WTTC 2005). WRI conducted a financial analysis of reef-related tourism, including net revenues from all reef-related activities, accommodation, and other spending on reef-related days. In addition, the study drew on a local-use survey to estimate recreational use of the reefs and coralline beaches by local residents each year. In total, coral reef-associated tourism and recreation contributes an estimated US$43.5 million to the national economy per year.

Revenues from reef-associated fisheries tend to be dwarfed by tourism in the Caribbean, but fishing is an important cultural tradition, safety net, and livelihood for many people. Coastal fishing communities are often among the most vulnerable groups to degradation of the ecosystem, as they may have fewer income alternatives. A financial analysis of reef-related fisheries in Tobago found that annual economic benefits are between US$0.8–1.3 million (0.7–1.1 million).

**The role of coral reefs in protecting the shoreline**

As part of this valuation effort, WRI developed an innovative method for evaluating the shoreline protection services provided by coral reefs. By integrating data on coastal characteristics, storm events, and coral reef location and type into a Geographic Information System, we are able to evaluate the role of coral reefs in maintaining the
stability of a country’s shoreline. In Tobago (Fig. 17.B.1), the relative reef contribution is zero in areas not protected by a coral reef, and ranges from 27% where the shoreline has relatively good protection due to other factors, to 42% where the shoreline would be most vulnerable without the reef. The relative share of protection provided by coral reefs is particularly high behind the Buccoo Reef in southwest Tobago, as well as along several portions of the windward coast. After assessing the relative protection provided by coral reefs, we integrate property values for vulnerable areas to arrive at an estimate of US$18–33 million in “potentially avoided damages” per year.

Policy relevance

This type of valuation produces a picture of the current estimated value of these three services. The method has the advantage of being simple, replicable, and transparent, and it is a useful exercise for drawing attention to reef-related benefits that are often undervalued or unnoticed. Even ballpark values help to support an economic case for including these types of ecosystem services in decision-making processes. Going forward, policy-makers in many Caribbean countries may find it worthwhile to invest in economic valuation to support decision-making, including conducting cost-benefit analyses and assessing the effects of coral reef degradation on the value of these services over time.

Working with local partners, WRI has tied the economic findings to some clear opportunities for improved coastal management in Tobago. For instance, the Buccoo Reef Marine Park (BRMP) in the southwest of the country is a cornerstone of the tourism industry—60% of international visitors take trips into the park—and provides significant coastal protection to a heavily developed and low-lying section of the island. Applying the same methods as at the national level but looking over a 25-year period, we estimate that tourism associated with BRMP contributes

![Shoreline Protection by Coral Reefs—Relative Reef Contribution](image)
and are responsible for approximately 40% of global net primary productivity (Schlesinger 1991; Melillo et al. 1993). The oceans are home to vast reservoirs of genetic and ecological diversity, arguably the most fundamental of supporting services as it is directly linked to the rate of evolution and therefore the ability to adapt to a changing climate (Pergams and Kareiva, in press).

The valuation of marine ecosystem services lags behind efforts aimed at terrestrial systems, although coastal wetlands (Batie and Wilson 1978; Lynne et al. 1981; Farber 1988; Bell 1989), coral reefs (Spurgeon 1992; Moberg and Folke 1999; Cesar 2000; Brander et al. 2007), and mangroves (Bennett and Reynolds 1993; Gilbert and Janssen 1998; Ronnback 1999; Ruitenbeek 1994; Barbier 2000; Sathirathai and Barbier 2001; Barbier 2003, 2007; Barbier et al. 2008) are notable exceptions. Marine and coastal ecosystem services have been included in a few comprehensive valuation exercises. Costanza et al. (1997) used a (mostly) benefits-transfer approach, applying estimates of ecosystem service values for specific terrestrial and marine habitats to extrapolate the global value of ecosystem services. Without careful matching of sites to ensure that the benefits can and should be transferred, however, this approach can be misleading (Plummer 2009). One of the most interesting discussions of ecosystem service valuation in the marine environment entails four case studies that demonstrate how valuing a suite of ecosystem services has the potential to inform decision-making in the Swedish coastal zone (Soderqvist et al. 2005). One of these case studies explores the costs (increased water treatment and reduced fertilizer use) and benefits (recreational and other cultural benefits) of improved water quality in the Stockholm Archipelago. Recent examinations of shoreline stabilization and trade-offs with aquaculture are illustrative of a general growing interest in services from coastal environments (Box 17.1, Barbier et al. 2008).

17.3 Mapping and modeling the flow of marine ecosystem services: a case study of Puget Sound

Ecosystem services are a useful currency for cost-benefit analyses or assessments of the trade-offs among alternative strategies for achieving multiple objectives in marine systems. This is especially true when those ecosystem objectives explicitly include human well-being in addition to traditional conservation goals, which is the situation in Washington’s Puget Sound region. In this section, we present a small step forward in applying the concept of quantifying dynamic flows of ecosystem services to the management of Puget Sound. The Puget Sound ecosystem in Washington State is home to 3.8 million people encompassed in a 42 000-km² basin, including temperate-latitude lands and rivers from the crests of the Cascade and Olympic mountains through a deep, fjord-type estuary to the Pacific Ocean. The region’s marine environment produces basic provisioning services such as commercial and tribal subsistence fisheries for salmon (Oncorhynchus spp.) and other species, as well as clam, oyster, crab, and other shellfish harvests. It provides regulating services as global as the carbon cycle, and as local as waste treatment through the breakdown of PAHs (polycyclic...
aromatic hydrocarbons) and PCBs (polychlorinated biphenyls) by eelgrass (Huesemann et al. 2009). It offers numerous cultural services through bird and whale watching, recreational fishing, water recreation, educational opportunities, and simply the human value placed on the existence of the region’s biodiversity. Puget Sound also provides a rich cultural heritage for native Indian tribes. And underlying all of these are basic supporting services such as primary production and the provision of habitat for the Pacific Northwest icons salmon and orcas (Orcinus orca).

Using Puget Sound as a case study is motivated by the region’s move toward an ecosystem-level management approach. In 2007, the Washington State Legislature mandated formation of a new State agency guided by a public-private council—the Puget Sound Partnership (Partnership)—whose charge is to recover the ecosystem by 2020. The Partnership’s governance structure and mandate for ecosystem recovery presents an opportunity to apply principles from ecological theory and the science of ecosystem services to help prioritize management actions for Puget Sound. The Partnership recognizes that ecosystem recovery will require changes in the way local, state, Federal and tribal governments act and—just as importantly—changes in choices human residents make about how they commute to work, where they buy their food, homes, and so forth (Puget Sound Partnership 2006). To meet these challenges, the Partnership has adopted a system-wide approach to restoring the ecosystem, and they have explicitly defined their multiple objectives in terms of what ecosystem services people in the region care the most about (Puget Sound Partnership 2008).

Identifying these public values is an essential step toward making an ecosystem services framework of practical use. In Puget Sound, a diverse group of stakeholders including those from fisheries and aquaculture, tourism, ports and shipping, cities, counties, tribal governments, environmental interests, agriculture, forestry, homebuilding, and business sectors were interviewed to identify those services they believe to be “most important.” The interviewers first educated the participants about the concept of ecosystem services and offered them a list of 24 services translated from the Millennium Ecosystem Assessment into locally relevant terminology. Across 12 different sectors, there was broad agreement that water quantity and water regulation, recreation and ecotourism, and ethical and existence values were of the utmost importance; capture fisheries, aquaculture, water purification and waste treatment also ranked highly (Iceland et al. 2008). Trade-offs are likely to occur among services even in this short list of valued ecosystem benefits. Representing outcomes of management choices in terms of multiple benefits, in currencies related to human well being, has promise for engaging a broader spectrum of the public in charting a path forward.

An early focus of the Partnership’s effort is nearshore habitats. This builds on the work of a number of previous planning efforts in Puget Sound, which identified specific actions aimed at either protecting existing nearshore habitats or restoring degraded areas to provide improved function for species, habitat maintenance, or human access (Shared Strategy 2007; Puget Sound Nearshore Ecosystem Restoration Project 2008; Alliance for Puget Sound Shoreslines 2008). These nearshore recovery schemes in Puget Sound have broadly similar objectives in their common desire to increase the amount of functioning nearshore habitat.

In the remainder of this chapter, we focus on the suite of ecosystem services that nearshore habitats produce and support in Puget Sound, and how those services could change in response to a set of possible management actions. To illustrate this approach, we quantify the outcomes of nearshore protection or restoration for three distinct kinds of services that flow from an important foundation species—eelgrass (Zostera marina): (1) carbon sequestration for climate regulation (a regulating service with global reach), (2) marine commercial harvest (a provisioning service), and (3) non-consumptive values (recreation and existence values) associated with species that belong to the Puget Sound food web. We examine how changes in ecosystem services are created by changes in eelgrass itself (carbon sequestration) and how changes in eelgrass produce changes in services provided through higher levels of the food web (harvest, recreation, and existence values).
17.3.1 Eelgrass

Eelgrass (*Zostera marina*) is a widely distributed, clonal seagrass that forms large, often monospecific stands in shallow temperate estuaries worldwide. Much of the vegetative biomass of eelgrass is below the surface of sediments and the above-ground biomass tends to be highly seasonal. Ecosystem services attributed to seagrass beds include the sequestration of carbon, the provision of habitat for fish and invertebrates, and the control of erosion through sediment stabilization (Williams and Heck 2001). Threats to eelgrass in Puget Sound are similar to those facing this habitat type elsewhere, including mechanical damage (such as through dredging and anchoring), eutrophication, some aquaculture practices, siltation, coastal construction, invasions by non-native species, alterations to coastal food webs, and climate change (Williams and Heck 2001; Duarte 2002; Bando 2006). Eelgrass beds currently occur along approximately 37% of the coast of Puget Sound, where they provide habitat for mobile organisms such as crabs and small fishes and feeding habitat for larger consumers such as seabirds, salmon, and marine mammals (National Marine Fisheries Service 2007). They also provide spawning and rearing habitat for Pacific herring (*Clupea pallasi*), a key species in the regional food web (Penttila 2007; National Marine Fisheries Service 2007).

To evaluate policy scenarios for the nearshore, we ask how management actions are likely to affect eelgrass, and then how multiple ecosystem services provided by eelgrass are likely to change as a result. We estimate ecosystem services using simple approaches for the sake of illustration, and do not include ecological nuances such as spatial or temporal variation in their production and delivery. Also, it is important to note that since we are examining changes in flows of services that are likely to result from changes in a foundation species, we focus on bottom-up effects; future scenario work will include the examination of top-down effects such as changes in harvest of key fish species.

We know from previous work that the total area of eelgrass in Puget Sound has declined from historical levels (Thom and Hallum 1990). Although the reasons for this decline are not well understood (Thom and Albright 1990; Thom and Hallum 1990), we assume for the sake of illustration that the Partnership can identify and implement policies capable of protecting eelgrass and halting this decline. In addition, we examine other policies aimed at restoring eelgrass in areas where it used to occur. To assess the potential for restoration, we built a spatially explicit habitat suitability model for eelgrass in Puget Sound, with the aim of identifying locations in which eelgrass has the potential to grow but where its current status is unknown (Figure 17.1; Plate 13, Davies et al., in preparation). Our model suggests that an additional 36,877 ha of Puget Sound’s benthic area has the potential to be occupied by eelgrass. If even half of this were to be occupied in the future due to restoration projects, it would represent nearly a doubling of the current area of eelgrass and would yield area similar to that estimated to be present historically (41,239 ha compared to 47,328 ha summarized from Thom and Hallum 1990).

17.3.2 How do changes in eelgrass habitats affect carbon storage and sequestration?

Estimates of carbon storage and sequestration in marine systems are rare. For example, a marine analog does not exist that is similar to the look-up tables for carbon storage and sequestration values for various land use/land cover categories available in terrestrial systems (Intergovernmental Panel on Climate Change 2006). Our approach to estimating the ecosystem service value of eelgrass carbon sequestration is similar to that of Chapter 7 (this book) and is summarized in Table 17.2a. We first estimate the amount of carbon stored in eelgrass biomass and soils. We then estimate the rate of carbon sequestration for eelgrass in Puget Sound, noting that carbon also flows through eelgrass to be consumed by herbivores, decomposed within the system, and exported from the system. These two estimates yield ecosystem service values for the scenarios examined. In the end, our approach illustrates how changes in eelgrass habitats result in changes in the amounts and values of carbon storage and sequestration.

17.3.2.1 Carbon storage

Our estimates of carbon storage in Puget Sound eelgrass beds and sediments range from 1–6.3 TgC (Table 17.2a). The amount of C in the sediment pool
Figure 17.1  A map of Puget Sound showing areas our model predicts suitable for eelgrass beds (green). Inset maps show higher detail; orange represents currently mapped eelgrass from the NOAA Essential Fish Habitat data (TerraLogic GIS Inc. 2004). (See Plate 13.)
Table 17.2a  
Summary of methods used for estimating how changes in ecosystem structure and function give rise to changes in services provided for carbon storage and sequestration, commercial fisheries, and food web support

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Estimation approach</th>
<th>Parameter values</th>
<th>Key assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Carbon storage by eelgrass for climate regulation</td>
<td>[ C = (A^*C_B + C_S) ]</td>
<td>( A = 21,140 ) ha</td>
<td>40% of eelgrass biomass is ( C ); ( C ) in soil is attributed to eelgrass stabilizing and trapping sediments and preventing decomposition.</td>
</tr>
<tr>
<td></td>
<td>( C_B = 0.03–0.4 ) ( \text{MgC/ha} )</td>
<td>( C_B = 0.03–0.4 ) ( \text{MgC/ha} )</td>
<td></td>
</tr>
<tr>
<td></td>
<td>( C_S = 50–300 ) ( \text{MgC/ha} )</td>
<td>( C_S = 50–300 ) ( \text{MgC/ha} )</td>
<td></td>
</tr>
<tr>
<td>Carbon sequestration by eelgrass for climate regulation</td>
<td>[ \Delta C = NPP<em>S</em>A ]</td>
<td>( NPP = 300–600 ) ( \text{gC/m}^2/\text{yr} )</td>
<td>Carbon exported to other systems (including the deep sea) has the potential to be sequestered; but because its ultimate fate is unknown it is not considered further here.</td>
</tr>
<tr>
<td></td>
<td>( S = 5–15% )</td>
<td>( S = 5–15% )</td>
<td></td>
</tr>
<tr>
<td></td>
<td>( A = 21,140 ) ( \text{ha} )</td>
<td>( A = 21,140 ) ( \text{ha} )</td>
<td></td>
</tr>
<tr>
<td>Commercial fisheries for food: current</td>
<td>Observed landed biomass by species for commercial fisheries in different marine sub-basins</td>
<td>Pounds per species/year landed in Puget Sound(^6)</td>
<td>Eelgrass affects herring biomass through egg survival; food web responses to changes in eelgrass are mediated through herring</td>
</tr>
<tr>
<td>Food web support mediated through eelgrass-herring interaction</td>
<td>EwE food web model(^7)</td>
<td>Sigmoid mediation function(^7)</td>
<td>Eelgrass affects herring biomass through egg survival; food web responses to changes in eelgrass are mediated through herring</td>
</tr>
<tr>
<td>Wildlife viewing and existence mediated through eelgrass-herring interaction</td>
<td>EwE food web model(^7)</td>
<td>Sigmoid mediation function(^7)</td>
<td>Eelgrass affects herring biomass through egg survival; food web responses to changes in eelgrass are mediated through herring</td>
</tr>
</tbody>
</table>

---

2. In one subtidal Puget Sound meadow Nelson and Waaland (1997) estimated annual above- and below-ground eelgrass biomass to average 256.3 \( \text{g dw/m}^2 \) (seasonal range: 72.2 \( \text{g dw/m}^2 \) in January to 445.0 \( \text{g dw/m}^2 \) in July). These are similar to Webber et al. (1987) from another Puget Sound location. Yang et al. (unpublished data) surveyed 17 sites around the sound in the spring and found above- and below-ground biomass to range 17–217 \( \text{g dw/m}^2 \). Because we are interested in estimating the \( C \) in relatively steady-state pools, we used winter biomass and chose a range 8–100 \( \text{g dw/m}^2 \).
3. Eelgrass sediments in Rhode Island have been characterized as having up to 300 \( \text{Mg C/ha} \) (Payne 2007). Jesperson and Osher (2007) found soils to a depth of one meter in an estuary in Maine to average 136 \( \text{Mg C/ha} \) with a range for different (generally unvegetated) habitats of 67–177 \( \text{Mg C/ha} \). We used a range of 50–300 \( \text{Mg C/ha} \) for the sediment \( C \) estimates. To put this in context, the global average for wetland soils is 720 \( \text{Mg C/ha} \) (US Department of Energy 1999) and Pacific Northwest old-growth forest soils are estimated to hold 30–400 \( \text{Mg C/ha} \) (Homann et al. 2004). For comparison, Pacific Northwest forests have been estimated to store 180 \( \text{Mg C/ha} \) above-ground (Lippke et al. 2003).
4. Globally, Mann (1982) estimated the NPP of coastal systems to range 300–1000 \( \text{g C/m}^2/\text{yr} \). Duarte and Cebrian (1996) estimated seagrass NPP to be 548 \( \text{g C/m}^2/\text{yr} \); Mateo et al. (2006) estimated it to be 817 \( \text{g C/m}^2/\text{yr} \). Estimates for Z. marina in Europe and Asia range 620–3 600 \( \text{g C/m}^2/\text{yr} \) (summarized by Stevenson 1988). Estimates of NPP of Z. marina in Alaska and Oregon are 1000–1500 and 316–450 \( \text{g C/m}^2/\text{yr} \), respectively (McRoy 1974; Kentula and McIntire 1986). An estimate of NPP for above-ground Z. marina and epiphytes at one location in Puget Sound is 344 \( \text{g C/m}^2/\text{yr} \) (Thorn 1990). Thorn (unpublished data) used an estimate of 600 \( \text{g C/m}^2/\text{yr} \) in Puget Sound for above- and below-ground biomass.
5. A carbon budget for generalized seagrass systems estimated that 15.9% of NPP is stored in sediments (Duarte and Cebrian 1996); because most studies have been conducted on a tropical genus that forms large mats of organic material, we assumed that 15% was an upper bound for Z. marina.
6. PacFIN (Pacific Fisheries Information Network), unpublished data.
7. See the text for description of the EwE model. We varied the shape of the mediation function from nearly linear to steeply sigmoid; as model outputs were qualitatively similar across all steepness terms, we discuss the results for an intermediate function.
Table 17.2b  Summary of methods for estimating how changes in ecosystem services result in changes in their value for carbon storage and sequestration, commercial fisheries, and food web support

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Estimation approach</th>
<th>Parameter values</th>
<th>Key assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Carbon storage and sequestration by eelgrass for climate regulation</td>
<td>Eelgrass protection: Reduction in expected damage from climate change through carbon storage and sequestration. Estimate difference in C stocks with and without eelgrass protection. 1, 2, 3</td>
<td>$T = 50$</td>
<td>Eelgrass is mature; sediment C losses when eelgrass biomass is lost span 3 periods: An initial period before eelgrass is lost in which sequestration continues; a second period in which eelgrass sediment carbon is lost (at a constant rate); and a third period in which sediment carbon remains stable at its minimum level.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>$C_s = 0.21$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$C_s = 175$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$\Delta C_s = 0$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$\Delta C_s = 0.525$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$C_{min} = 43.75$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$t_1 = 10$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$t_2 = 20$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$p = $25$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$r = 3%$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$c = 3%$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$T = Years of carbon sequestration$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$C_B = 0$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$C_S = 0$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$\Delta C_B = 0.0432$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$\Delta C_S = 0.525$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$T_B = 5$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$\pi = 0.5$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$p = $25$</td>
<td>C stored and sequestered in area to be restored is 0.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>$r = 3%$</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>$c = 3%$</td>
<td></td>
</tr>
</tbody>
</table>

$T = Years of carbon sequestration$

$C_s = Carbon stock for biomass (MgC/ha)$

$C_B = Carbon stock for sediments (MgC/ha)$

$\Delta C_s = Annual carbon sequestration for biomass (MgC/ha/yr)$

$\Delta C_B = Annual carbon sequestration for sediments (MgC/ha/yr)$

$T_B = Years for restored eelgrass to reach "maturity"$

$\pi = Probability of successful restoration$

$p = Social value (\$/MgC)$
<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Estimation approach</th>
<th>Parameter values</th>
<th>Key assumptions</th>
</tr>
</thead>
<tbody>
<tr>
<td>Commercial fisheries for food: current</td>
<td>Net revenues by species for commercial fisheries in different marine sub-basins</td>
<td>Pounds and dollars per species/year landed in each marine sub-basin</td>
<td>Harvest rates for all species do not change; non-trophic relationships between eelgrass and other species (e.g., Chinook salmon and Dungeness crab) are not examined. Non-commercial value is related to population size (lbs are used as a proxy for value); non-trophic relationships between eelgrass and other species (e.g., Chinook salmon) are not examined.</td>
</tr>
<tr>
<td>Commercial fisheries for food: mediated through eelgrass-herring interaction</td>
<td>Use food web model to examine changes in commercial harvest due to changes in eelgrass biomass</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wildlife viewing and existence mediated through eelgrass-herring interaction</td>
<td>Use food web model to examine changes in biomass of species groups due to changes in eelgrass biomass</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\( r = \text{Social discount rate} \)
\( c = \text{Carbon discount rate} \)

1. With protection, the amount of carbon in year \( t \) is

\[
C_{sw}(t) = C_r + C_t + t \Delta C_r.
\]

2. Without protection, the amount of carbon in year \( t \) follows a step function:

\[
C_{wo}(t) = \begin{cases} 
C_r + C_t + t \Delta C_r & \text{if } t \leq t_1 \\
C_{wo, t} & \text{if } t > t_1, \end{cases}
\]

3. The economic value of eelgrass protection is derived by first considering the difference between the carbon stock in each year with and without protection:

\[
V(\text{Protection}) = \sum_{t=1}^{T} \frac{\rho \Delta C_{sw}(t) - \rho \Delta C_{wo}(t)}{(1+c)^t}
\]

4. With restoration, the amount of carbon in year \( t \) is

\[
C(t) = \Delta C_r + \Delta C_t, \quad t < T_r
\]

\[
= \Delta C_r, \quad T_r \leq t \leq T
\]

5. Without successful restoration, the amount of carbon in any year is zero, and so the expected value of eelgrass restoration is

\[
V(\text{Restoration}) = \rho \sum_{t=1}^{T} \frac{\Delta C_r}{(1+c)^t}
\]

6. PacFIN (Pacific Fisheries Information Network), unpublished data.

7. See the text for a description of the EwE model.
dwarfs that in the biomass pool (such that the biomass pool is truly negligible). In comparison, total US forest carbon stocks are estimated to be in the range of 40000–50000 TgC. Pacific Northwest (western OR and WA) forests are estimated to contain approximately 351 TgC (US Environmental Protection Agency 2007).

17.3.2.2 Carbon sequestration
Seagrasses stabilize sediments, slow water motion, and cause the deposition of organic matter from the water column (Gacia and Duarte 1999; Gacia et al. 1999). Below the sediment surface, anoxia and light-limitation inhibit microbial processing and photodegradation (Jesperson and Osher 2007), allowing for the build-up of C. Soil carbon generally has long residence times—particularly when submerged—and is therefore considered “sequestered carbon” (Wang and Hseih 2002). Despite ideal conditions for production and preservation of organic matter, the C-sequestration capacity of the soils of coastal ecosystems has been understudied (Chmura et al. 2003; Thom et al. 2003; Jesperson and Osher 2007).

Our initial estimate of a sequestration rate in Puget Sound is 3171–19026 Mg C/yr, or 11.627–69.762 Mg CO₂/yr (Table 17.2a). This represents 0.02–0.1% of the emissions of Washington State, 0.06–0.36% of the emissions of King County (Seattle’s home), and up to 72% of the annual emissions of all transit busses in King County (King County 2007). For comparison, the carbon contained in all US forests offset approximately 10% of total US CO₂ emissions in 2005 (Woodbury et al. 2007).

If a restoration policy is being pursued, particular habitat types would be changing to eelgrass from a previous habitat type, and the original state would have had its own C-storage/sequestration values. Similarly, if eelgrass habitat is being lost, it is being replaced with another habitat type, and a similar comparison could be made. This makes marginal changes impossible to calculate without going through the same exercise for all habitat types. Among possible habitat types, however, we expect eelgrass to have the greatest capacity for carbon storage and sequestration, compared to other non-vegetated intertidal and shallow subtidal habitats such as rocky reefs, cobble, mud-, or sand-flats.

17.3.2.3 Valuing ecosystem service value changes for carbon
The ecosystem service value of carbon storage and sequestration is based on the reduction in the expected damage from climate change. Increasing levels of carbon dioxide and other greenhouse gases are linked to harmful changes in temperature and other aspects of climate. Controlling carbon dioxide by sequestering carbon therefore mitigates those harmful effects, which counts as an economic benefit. This value is enjoyed by society at large, and so it is referred to as the social value of carbon storage and sequestration.

For eelgrass, this ecosystem service value can be generated either by protecting current eelgrass or investing in eelgrass restoration, as described above. Protection provides value if existing eelgrass areas are threatened and the projected amount without protection decreases over time; restoration provides value when eelgrass would otherwise decline or remain stable in the future. In either case, the economic value is based on the difference in carbon stocks over time for two scenarios, one with the appropriate action (protection or restoration) and one without that action. Thus, it is important to understand what form of carbon sequestration and storage (if any) would either replace eelgrass (for the case of protection) or be replaced by eelgrass (in the case of restoration). Calculating this economic value is relatively straightforward (Chapter 7), but settling on the values for some of the parameters is fraught with controversy (Nordhaus 2007; Weitzman 2007). For the purposes of this chapter, we pick values merely to illustrate how the calculation and resulting value depends on the scenarios described above (Table 17.2b).

Based on the methods outlined in Table 17.2b, the social value of eelgrass protection is much higher than that of restoration: $1 496 to 4 585 versus $104 ha⁻¹. The range in values for protection reflect different assumptions about the loss of carbon from sediments, with the high value representing an assumed total loss of stored carbon when eelgrass is left unprotected and the low value resulting from a ten-year lag before carbon is released, a ten-year carbon release period, and a 25% minimum carbon stock that remains in the sediments without eelgrass. The large disparity is due in large part to the
Figure 17.2  A map of the Puget Sound Partnership's action areas showing the distribution of (a) landings (in UK£) and (b) revenue (in US$) of farmed and wild seafood from 1998 to 2007. (See Plate 14.)
assumed loss of stored carbon when eelgrass is left unprotected. If leaving eelgrass unprotected does not produce a significant loss of stored carbon in the sediments, the value of protection and its advantage over restoration is diminished accordingly.

17.3.3 Marine harvest and non-consumptive values

Puget Sound’s living marine resources, though depleted relative to historic times, remain a bountiful source of provisioning and other ecosystem services. Commercial fisheries harvest over 35 species of finfish and shellfish, and generate more than $50 million in annual revenue (Pacific States Marine Fisheries Commission, Pacific Coast Fisheries Information Network (PacFIN), unpublished data). Recreational harvest concentrates on Pacific salmon and steelhead (Oncorhynchus mykiss) but also includes shellfish such as Dungeness crab (Cancer magister) and butter clams (Saxidomus giganteus) (Washington State Department of Fish and Wildlife (WDFW), unpublished data). Puget Sound Indian tribes enjoy a rich tradition of ceremonial harvest. Aquaculture uses the ecological functioning of Puget Sound to produce more than $30 million in annual revenue for shellfish and almost $20 million for Atlantic salmon (Salmo salar) (WDFW, unpublished data). Non-consumptive activities, such as recreational whale and bird watching, also provide an important flow of services, while just the existence of some species produces value for Puget Sound residents and visitors.

In order to provide a perspective with which to gauge the effects of changes in the ecosystem, we tallied the commercial harvest coming from Puget Sound in biological (lbs.) and monetary ($) units (PacFIN; Figures 17.2a, b; Plate 14). This snapshot of the seafood provisioning service allows us to examine where particular types of harvest are highest. For example, shellfish, which are particularly lucrative, are predominately produced in southern Puget Sound.

The mobility of fish and their use of multiple habitat types necessitates a food web-based modeling approach rather than a habitat-based one. To understand how changes in nearshore environments and eelgrass are likely to affect changes in the food web-based ecosystem service flows, we focus on the link between eelgrass, a foundation species, and one other species with wide-ranging food web interactions, Pacific herring.

17.3.3.1 How might changes in eelgrass habitats affect marine harvest and non-consumptive values?

In order to begin to understand how changes in nearshore environments are likely to affect changes in the flows of harvest and other services, we examined the habitat associations of the top 25 species harvested in the sound (including: geoducks (Panopea abrupta), salmon, Dungeness crab, and oysters), and categorized their dependence on nearshore habitats. Only 5 of these species (spiny dogfish, Squalus acanthias, and 4 salmon—steelhead; sockeye (Oncorhynchus nerka), coho (Oncorhynchus kisutch), and pink (Oncorhynchus gorbuscha)) did not rely on nearshore habitats for at least one part of their life cycle. Thus, harvest levels of most species in the top 25 are likely to be sensitive to changes in nearshore habitats, but further modeling is necessary to understand how.

Pacific herring are a key food web species that interacts with eelgrass—they aggregate in the nearshore prior to reproducing then spawn in shallow water, usually on submerged vegetation (eelgrass or algae). Submerged vegetation provides spawning substrate, food resources, cover, and nursery habitat (Thayer and Phillips 1977; Dean et al. 2000; Penttila 2007). Survivorship of eggs is higher with lower spawn density (Galkina 1971; Taylor 1971). To survive, planktonic larvae must have sufficient supplies of microplankton; blooms of which are believed to be earlier, more dense and more consistent in sheltered bays (Penttila 2007). The survival of larval herring (determined particularly by food availability and predation) is thought to have a significant impact on the future abundance of the year-class (Alderdice and Hourston 1985). Juveniles spend several months inshore before moving into deeper waters (Penttila 2007). Herring are important prey to seabirds, crabs, salmon, marine mammals, and numerous other groups (Haegele 1993a, b; Penttila 2007). Given the strong connection between herring and nearshore habitat, we focus here on the consequences of how changes in
nearshore habitat give rise to changes in herring, and how such effects can propagate through the food web.

17.3.3.2 Puget Sound food web model
Biomass dynamics of eelgrass and herring take place in the context of a broader community of interacting species, and resulting feedbacks within the food web are difficult to anticipate without the benefit of models. We used the Ecopath with Ecosim (EwE; Christensen and Walters 2004) software to construct a food web model for the central basin of Puget Sound (Figure 17.3a; Plate 15; Harvey et al. 2010). EwE models trophically and reproductively link biomass pools using a mass-balance modeling approach that satisfies two master equations describing production (as a function of catch, predation, migration, and biomass) and consumption (as a function of production, respiration, and unassimilated food). An initial mass-balanced snapshot of the ecosystem can then be used to explore dynamic simulations by expressing biomass flux rates among pools through time. The model of Puget Sound’s Central Basin (Harvey et al. 2010) has functional groups ranging from primary producers to marine mammals and seabirds, as well as several fisheries. The model results we present below are preliminary outcomes that illustrate the complex and often unforeseen nature of community responses to perturbations (Christensen and Walters 2004).

Manipulating eelgrass production in EwE has negligible effects on the food web through consumption—a result that reflects our current understanding of eelgrass as a relatively unimportant direct food source (Mumford 2007). In contrast, non-trophic effects of eelgrass—such as habitat provisioning—are known to be very important (Thayer and Phillips 1977; Orth et al. 1984; Hosack et al. 2006; Mumford 2007). Such effects have important positive effects on other species and can be reflected in EwE through density-dependent mediation functions (Ainsworth et al. 2008). Mediation functions quantitatively link the vulnerability of a group (in this case, herring eggs) to the biomass of a mediating group (in this case, eelgrass): in other words, the less eelgrass is present, the more vulnerable herring eggs are to their predators. This is but one of numerous mechanisms by which changes in eelgrass could lead to changes in herring populations. Here we present only indirect effects that act through herring (Figure 17.3b; Plate 15) and focus on species and/or groups whose relationship with eelgrass is either trophic (i.e., they consume it) or is mediated through direct or indirect trophic interactions with herring.

We simulated a 50% decrease, a 50% increase, and a doubling of eelgrass biomass, and linked this to herring egg vulnerability. Depending on eelgrass biomass, herring eggs became either more or less vulnerable to predation by several groups (ducks and brants, gulls, ratfish, Dungeness crabs, small nearshore fishes, and small crustacean omnivores). Increases in eelgrass biomass yielded increases in herring and in turn increases in harbor seals (Phoca vitulina, whose primary prey is herring), ducks and brants (consumers of eelgrass), and greenlings (consumers of small crustaceans who feed on herring eggs). Increases in eelgrass yielded decreases in gulls and terns, skates, gadoids, lingcod (Ophiodon elongatus), and numerous flatfish. Most of these decreases result from competition with increased herring populations. Skates, however, likely decline with increases in eelgrass and herring because their primary predators are harbor seals, a species that increases with eelgrass and herring. Results for decreases in eelgrass biomass generally mirrored those of increases.

17.3.3.3 Valuing commercial harvest and non-consumptive services from food web changes
The species in the Puget Sound food web model provide consumptive services that include commercial harvest, and non-consumptive services such as whale and bird watching and existence value (because of data limitations on recreational fishing values, we do not consider recreational harvest in this section). Of the functional groups in the Puget Sound food web model, 15 are harvested commercially (Table 17.3). By weight, the most important fisheries are salmon (sockeye; chum, Oncorhynchus keta; coho; and Chinook, Oncorhynchus tshawytscha, both wild and hatchery stocks), geoduck, Pacific herring, and Dungeness crab, in that order. By value, the same fisheries dominate but the geoduck’s high price per pound makes it the most economically
Figure 17.3  (a) The structure of the EwE food web model of the Central Basin of Puget Sound (without fisheries) and (b) a subset of the EwE food web model focusing on eelgrass and herring.
(a) Box size is proportional to standing stock biomass; line thickness is proportional to the flow of energy/material from the prey to the predator. Red colors represent detritus and the portion of the food web it supports, blues are benthic primary producers and those they support, and greens are phytoplankton and phytoplankton-supported groups. Consumers’ colors are a mix proportional to the amount of production that ultimately stems from those sources. In (b) dashed arrows indicate groups whose predation on herring eggs is mediated by the biomass of eelgrass. Colors are as those in (a). (See Plate 15.)
valuable commercial species ($5.4 million, average annual revenue, 2005–7; PacFIN data). This value is split about evenly between the northern and southern parts of the Central Basin. Salmon and herring harvest, valued annually at $3.8 million and $159.3 thousand respectively, however, occur predominantly in the southern part of the Central Basin. Fifteen functional groups in the food web, including orcas, seals, ducks, sea stars, and the three groups of wild salmon, arguably have non-consumptive economic values based on outdoor recreation or simply for their existence (Table 17.3).

How are these values affected by the changes in nearshore conditions we have modeled? In biological terms, the food web model results show that the abundance of some species increases while it decreases for others. An ecosystem service value framework provides us with a way of evaluating these trade-offs. Ideally, expressing values in a common metric (dollars) enables one to make a grand aggregation of all the changes, producing a bottom line in terms of how nearshore conditions determine Puget Sound ecosystem service values. Two primary roadblocks prevent such an aggregation at this point: (1) we lack complete data on these values—commercial species are relatively easy to value, non-consumptive value species are not; and (2) we have not modeled the non-trophic relationships between eelgrass and a number of important species (e.g., Chinook salmon, chum salmon, and Dungeness crab).

To address these two issues, we assume that the non-consumptive value of a species is related to population numbers, using pounds as a proxy metric for these values; and we limit our discussion below to species whose primary interaction with eelgrass is through direct consumption of eelgrass or is mediated through their interactions with herring (Table 17.3). Among these species is Pacific hake (or whiting), *Merluccius productus*, which is currently considered a “species of concern” by NOAA Fisheries. This status indicates some concern about the viability of the species but insufficient information is available to make a formal ruling (National Marine Fisheries Service 2004). We consider this species separately, then, because further decreases in its status might trigger additional legal protections. Other species that currently have legal protection under the ESA (i.e., Chinook and summer chum salmon, steelhead, orca) are not examined here because they neither consume eelgrass nor primarily interact with eelgrass indirectly through their interactions with herring.

For the limited set of commercial species identified in this way, the herring fishery is the most important (in pounds harvested and revenue). Assuming the harvest rates for all species do not change, total herring harvest responds positively to changes in eelgrass, approximately doubling as eelgrass ranges from 50 to 200% of its baseline level. In contrast, spiny dogfish harvest increases by about 24% over that range, and surf smelt harvest decreases by about 23%; however, harvest yields for both of these species are considerably less than that of herring. Over the modeled range of eelgrass changes and using the average prices for each fishery (2005–7; PacFIN), the total harvest revenue for this limited set of commercial species would

---

**Table 17.3** Groups with significant commercial and non-consumptive values in the Puget Sound food web

<table>
<thead>
<tr>
<th>Commercial harvest value</th>
<th>Non-consumptive value (group)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Geoducks</td>
<td>Transient orcas (cetacean)*</td>
</tr>
<tr>
<td>Sockeye and chum salmon</td>
<td>Resident orcas (cetacean)*</td>
</tr>
<tr>
<td>(wild &amp; hatchery)</td>
<td></td>
</tr>
<tr>
<td>Chinook and coho salmon</td>
<td>Porpoises (cetacean)*</td>
</tr>
<tr>
<td>(wild &amp; hatchery)</td>
<td></td>
</tr>
<tr>
<td>Dungeness crab</td>
<td>Gray whales (cetacean)*</td>
</tr>
<tr>
<td>Clams (various spp.)</td>
<td>Harbor seals (pinniped)*</td>
</tr>
<tr>
<td>Pacific herring*</td>
<td>Sea lions (pinniped)*</td>
</tr>
<tr>
<td>Shrimp (various spp.)</td>
<td>Gulls (bird)*</td>
</tr>
<tr>
<td>Sea cucumbers</td>
<td>Piscivorous diving birds (bird)*</td>
</tr>
<tr>
<td>Dogfish*</td>
<td>Murrelets (bird)*</td>
</tr>
<tr>
<td>Burrowing shrimp</td>
<td>Ducks and brants (bird)*</td>
</tr>
<tr>
<td>Wild pink salmon</td>
<td>Seastars (invertebrate)</td>
</tr>
<tr>
<td>Surf smelt*</td>
<td>Wild Chinook and coho salmon</td>
</tr>
<tr>
<td>Sea urchins</td>
<td>Wild pink salmon salmon</td>
</tr>
<tr>
<td></td>
<td>Wild sockeye and chum salmon</td>
</tr>
<tr>
<td></td>
<td>Pacific hake (species of concern)*</td>
</tr>
</tbody>
</table>

* Species whose primary interaction with eelgrass is mediated through their interactions with herring (or who directly consume eelgrass). The commercial list includes all species groups with more than $1000 annual harvest in 2003–7. The non-consumptive group is subjectively chosen to represent species humans care about. The “group” for species of non-consumptive values indicates assignments to taxonomic groups for analysis of responses to eelgrass and herring perturbations in the EwE model.
increase by 82% or $942,000 as eelgrass increases to 200% of its baseline.

For non-consumptive value species, the aggregate weight of this group is negatively related to increases in eelgrass biomass, decreasing by 15% across the range of eelgrass levels. Expressing their total value as a simple summation of pounds, however, implicitly assumes that these species have an equal per-lb economic value. Although data are not available to provide any guidance to differentiate these values, dividing the group into subgroups defined by taxonomy and legal status reveals potentially important differences (Figure 17.4). Pinnipeds have a strong positive relation with eelgrass biomass, birds and cetaceans have a very weak positive relation, and Pacific hake exhibits a negative relation.

These results are heavily qualified, of course, by the absence in our modeling to date of ecological relations between eelgrass and species other than herring. It is not yet possible to assess the overall direction of the change in values captured in the Puget Sound food web in response to changes in nearshore habitat conditions. Our point here, however, is not so much to “accurately” depict the ecology of central Puget Sound, as to illustrate some important issues for using ecosystem service values. Commercial fisheries harvests are one of the most straightforward and easily measured ecosystem service values, and so producing a credible “bottom line” for this ecosystem service is possible as long as there are credible food web models. The modeling also allows us to understand that trade-offs among individual fisheries are still possible, and so improvements in ecological conditions may not be universally supported. The same can be seen in the trade-offs among non-consumptive value species.

17.3.4 Suites of ecosystem services in space

Overlying carbon storage and sequestration services, and marine harvest and non-consumptive values is another way to consider spatial variation in ecosystem services. Herring spawn in twenty to twenty-one locations around Puget Sound and

![Figure 17.4](image_url)  
**Figure 17.4** EwE model results for taxonomic groupings of non-consumptive value species (see Table 17.3 for group membership). The eelgrass index is eelgrass biomass/initial eelgrass biomass so values to the right of 1 represent increases in eelgrass biomass from the original baseline and values to the left represent decreases.
observations during years of relatively high abundance suggest that they may expand their spawning activities adjacent to currently used meadows, rather than colonizing new beds (Penttila 2007). Therefore, eelgrass restoration, if undertaken, will likely produce more value in areas adjacent to documented herring spawning sites where the benefits of increased carbon storage and sequestration are most likely to be complemented by the benefits of increased herring spawn, increased herring populations, and associated benefits derived from the food web (e.g., Figure 17.5).

This example illustrates the need to consider ecosystem services en suite, rather than one-by-one. It might be, for example, that the most productive areas for eelgrass restoration in terms of carbon services are not adjacent to existing herring spawning locations. The question is then whether the additional services derived from herring populations are worth the lesser carbon services. An ecosystem service framework can answer this question easily if a common metric (e.g., dollars) is used to measure the value of both sets of services. Even absent that information, the framework can illustrate where trade-offs may exist among services. In this way, policy-makers gain an understanding of the nature and extent of such trade-offs and can better set priorities in accordance with public values.

17.4 Future directions

The analyses presented here represent an initial step in developing an ecosystem services framework to support ecosystem-based management in Puget Sound. In this final section, we sketch out additional steps that can move such a framework closer to fruition.

---

**Figure 17.5** A portion of the Kitsap peninsula in Central Puget Sound, showing current eelgrass beds (hatched), areas used by herring for spawning (stippling), and areas predicted to be suitable for eelgrass restoration (dark gray).
A more complete evaluation of protection or restoration strategies will incorporate spatial variation in carbon storage and sequestration or food web functions and assess specific locations in terms of their current or potential production of these benefits. This information can provide useful guidance for recovery of the Puget Sound nearshore by providing a map with sites categorized according to their likely ecosystem service benefits under protection (for currently intact sites) or restoration (for sites that are currently degraded but with high intrinsic potential) strategies.

Such an evaluation also needs to expand the modeling of marine ecosystem services beyond the current set covered. Incorporating more links between nearshore habitat conditions and the marine food web will allow us to investigate other potential trade-offs among the provisioning services of commercial fisheries for salmon and other finfish, clam, oyster, crab, and other shellfish harvests, as well as the numerous cultural services that include bird and whale watching and recreational fishing. Waste treatment through the breakdown of PAHs (polycyclic aromatic hydrocarbons) and PCBs (polychlorinated biphenyls) by eelgrass should be added to the list of marine ecosystem services included in the analysis.

It is also possible in Puget Sound to extend the ecosystem services approach to include upland activities and their associated ecosystem services—basin-wide maps exist of current provisioning of water yields, water retention for floods, water purification potential, carbon storage, and commercial values of working landscapes in watersheds (Aukema et al. 2009; Rogers and Cooke 2009). The Partnership is interested in understanding how those watershed-based ecosystem benefits affect nearshore services provided to inform how and where to encourage different land-use practices around the region.

Clearly, developing a framework for assessing marine ecosystem services useful to policy-makers is an ambitious undertaking. Marine systems lack the commonly available spatial data that inform assessments of terrestrial services. As a result, building an assessment toolkit for marine environments may always be reliant on a richer set of local data and models developed for a particular location. We are using lessons learned from working in the Puget Sound region to develop models for multiple marine ecosystem services. The Marine Initiative of the Natural Capital Project is developing a marine InVEST tool that, like its terrestrial counterpart, will be an ecosystem services scenario assessment tool for application in ecosystem-based management processes with diverse stakeholders and across multiple scales. Building on the success of InVEST on land, we will connect existing models through the land-sea interface to new and existing marine models.

Ultimately, quantifying, mapping, and valuing marine ecosystem services has the potential to fundamentally change the ways in which decisions about marine and coastal environments are made. Making explicit the connections between human activities in one sector and their effects on a broad range of other sectors forces decision-makers and the human communities they represent to think about whole ecosystems and to manage them accordingly. By making clear the life-sustaining services oceans and coasts provide, appropriately valuing marine natural capital can help human communities make better choices about how we use these treasured environments.

References


World Travel and Tourism Council (WTTC). (2005). Trinidad and Tobago: the impact of travel and tourism on jobs and the economy.