Valuation of Ecosystem Services from Shellfish Enhancement: A Review of the Literature

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1 Introduction

This report is an expanded version of a literature review prepared by Northern Economics, Inc. for the Pacific Shellfish Institute in 2009 under a NOAA National Marine Aquaculture Initiative Grant. As with the earlier version, the purpose of this updated report is to provide planners and decision makers an overview of the existing literature on the valuation of ecosystem services provided by shellfish enhancement.[1] In order to orient non-economists to economic valuation, the report also describes the economic concepts and methods that have been applied in this literature. The primary differences between the current literature review and previous one are a greater focus on aquaculture production, an expanded review of the valuation of water quality improvement benefits and an added discussion of best management practices and standards.

The types of shellfish this report is concerned with are the bivalve mollusks, including clams, mussels, and oysters.[2] These shellfish are found throughout the coastal United States; some of the species are native to the areas where they occur, while others have been deliberately or inadvertently introduced. The emphasis of the literature review is on socioeconomic and biological studies of U.S. shellfish resources; however, studies of shellfish enhancement projects outside the United States are also discussed.

Within the scientific literature there is growing recognition of the central role shellfish can play in the maintenance and stability of coastal ecosystems. For example, oysters have been labeled "keystone species" (or "cornerstone species") in selected marine environments (Isaacs et al. 2004). As a keystone supports an arch, keystone species support a complex community of species by performing a number of functions essential to the diverse array of species that surround them. There is also increasing recognition that some shellfish species may impact or control many ecological processes; so much so that they are included on the list of "ecosystem engineers"—organisms that physically, biologically or chemically modify the environment around them in ways that influence the health of other organisms (Jones et al. 1994). Many of the ecological functions and processes performed or affected by shellfish contribute to human well-being by providing a stream of valuable services over time. Such services are commonly referred to as "ecosystem services" (The Millennium Ecosystem Assessment 2005).[3]

Although there is increasing recognition that shellfish provide multiple ecosystem services, management of shellfish and their habitats for objectives beyond recreational and commercial harvest has not yet become widespread (Brumbaugh and Toropova 2008). The remaining sections of this chapter examine why many of the ecological services provided by shellfish are in short supply, and how the economic valuation of these services can help rectify this problem.

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[1] Following the definition provided by Caddy and Defeo (2003), the term "enhancement" is used here to mean any intervention that improves the productivity of a shellfish resource and renders the productive activity more sustainable. It includes shellfish restoration and aquaculture as well as management (i.e., the sustainable use of natural shellfish beds).

[2] The bivalve mollusks currently or historically cultured in the United States as food include oysters of several species (*Crassostrea virginica, C. gigas, C. ariakensis, C. sikamea, Ostrea lurida* and *O. edulis*), mussels (*Mytilus edulis, M. trossulus and M. galloprovincialis*), several venerid (family Veneridae) clams (*Mercenaria mercenaria, Protothaca staminea* and *Venerupis phillipinarum*), scallops (*Argopecten irradians*), geoducks (*Panopea generosa*), soft-shell clams (*Mya arenaria*), cockles (*Clinocardium nuttallii*), rock scallops (*Hinnites giganteus*), arks (*Anadara transversa* and *A. ovalis*) and razor clams (*Siliqua alta, S. costata* and *S. patula*) (National Research Council 2010).

[3] A number of authors (e.g., Boyd and Banzhaf 2007; Fisher and Turner 2008) have pointed out shortcomings in the way ecosystem services were defined and classified by The Millennium Ecosystem Assessment. While this report acknowledges these critiques, it accepts The Millennium Ecosystem Assessment's definition and classification scheme for simplicity's sake.

1.1 Why the Production of Ecosystem Services May Be Suboptimal

In his book *Economic Values and the Natural World*, the economist David W. Pearce noted: "If the Earth's resources were available in infinite quantities, and if they could be deployed at zero cost, there would be no economic problem. Everyone could have everything they wanted without compromising each other's or later generations' wants and needs. It would not be necessary to choose" (Pearce 1993). But resources are finite, and we do have to choose among the goods and services we derive from those resources. The trade-offs people make are based on their preferences about the goods in question. If the good is bought and sold in a market, the marginal value of the good to society is reflected in its market price. The price acts as a "signal" that leads to the optimal allocation of resources among competing uses and production of goods.[4]

However, most of the services provided by a healthy, functioning ecosystem are what economists call "pure public goods" (Costanza et al. 1997). Specifically, these services exhibit the public good characteristics of non-rivalry and non-excludability. In the context of ecosystem services, non-rivalry means that more than one person can enjoy the benefits of an intact ecosystem at the same time. Non-excludability means that it is difficult (costly) to prevent one individual from enjoying the benefits created by another individual's actions to protect and preserve an ecosystem.

A consequence of these two characteristics of public goods is that the goods are rarely exchanged through markets, and thus the market system fails to allocate and price the goods correctly (Perman et al. 2011). The lack of market prices is often interpreted as if the public goods have no value, a fact that in the case of ecosystems leads to overuse or excessive exploitation (or underinvestment in protection and restoration) even though they provide services that people find beneficial.

1.2 How Economic Valuation of Ecosystem Services Offers a Possible Solution

For an economist, the question of the value of ecosystem protection or restoration and the larger question of how much protection or restoration should occur are central to the understanding of whether society should spend money on ecosystem protection or restoration or some other important "good" (Hicks 2004). Economic valuation of ecosystem services can be defined as the process of expressing a value for these services in monetary terms. Estimating the economic value of resources is frequently an important element in the formation and institution of efforts to prevent the twin problems of overexploitation and under-provision of public goods (Isaacs et al. 2004).

Today, the identification and quantification of ecosystem values is not only possible, it is increasingly seen as essential for the efficient and rational allocation of environmental resources among competing social and political demands (National Research Council 2004). Once described and accounted for, ecosystem service values can be used to make comparisons between competing economic tradeoffs. Furthermore, by estimating and accounting for the economic value of ecosystem services, costs or benefits that otherwise would remain hidden are revealed and vital information that has often remained outside of the decision-making calculus at local, regional and national scales can be brought to the foreground of decision making. A good example is the population and development pressures that estuarine areas are now experiencing. These pressures raise significant challenges for planners and decision makers (Wilson and Farber 2008). Communities must often choose between the myriad services provided by intact functioning ecosystems and competing uses of the coastal environment. To choose from these competing options, it is important to know what ecosystem services will be

[4] These economic principles truly apply only in a "perfect" market, i.e., one that is highly competitive, with many buyers and sellers all of whom have complete knowledge about the market. In reality, markets are not perfect, and therefore prices may reflect values only imperfectly (Gittinger 1982).

affected by coastal development or management and how these services create value for different members of society (Wilson and Farber 2008). Moreover, economic valuation of these ecosystem services gives them a common currency with marketed goods (e.g., beachfront condominiums constructed on landfill) so that ecosystems are not underrepresented in decision making (Carson and Bergstrom 2003).

While most of the ecosystem services provided by shellfish enhancement are not sold in markets and, therefore, not priced, there are exceptions, the principal one being the commercial harvest of shellfish for food. The relative importance of this ecosystem service may be relatively minor. An assessment of shellfish meetings over five years conducted by Luckenbach et al. (2005) revealed more than 300 presentations related to oyster restoration, with fewer than 25 percent focused solely on the protection of oyster fisheries. However, in the absence of information on the value of unpriced ecosystem services, the default measure of project success tends to be fisheries-based metrics such as harvest of market-sized oysters. The results are often disappointing due, in part, to a mismatch between the scale of restoration and measured outcomes. Citing a U.S. Army Corps of Engineers report, Brumbaugh et al. (2007) note that even relatively small shellfish restoration projects can be costly (e.g., > $100,000 per acre for restored oyster reef) relative to the value of oyster landings measured on the same area. However, valuation of non-fishery-related ecosystem services may reveal that the cost of restoration actions is justified even though it could not be justified by considering fishery benefits alone.

Valuation of ecosystem services is also important because not all services may be compatible with each other. A choice is often required among them, and this choice will dictate the total value of the ecosystem. For wild stocks of oysters to be of value as a food source, they must be harvested, thus affecting their ability to provide some other ecosystem services, such as providing habitat for themselves or other valued species, reducing shoreline erosion and contributing to water quality through their filtering capability (Sequeira et al. 2008). The valuation of the full array of services would enable planners and decision makers to better understand tradeoffs inherent in areas managed for harvest versus other uses (Brumbaugh and Toropova 2008). Lipton et al. (2006) note that the failure to acknowledge the total costs of oyster harvest in Chesapeake Bay management strategies contributed to the long term decline of the resource and, consequently, its asset value. Today, oysters in Chesapeake Bay have been reduced to one percent of their former abundance (King and McGraw 2004). Of course, a shellfish aquaculture operation can replant once harvest is complete and thereby reestablish the ecosystem services generated by the operation.

In short, economic valuation is a critical factor in ensuring that ecosystems will be maintained in a sustainable manner. Putting a dollar value on ecosystem services does not devalue them—on the contrary, it gives these services a currency so that they can be compared with the economic values of activities that may compromise them (Carson and Bergstrom 2003; National Research Council 2004). Furthermore, the results of economic valuation can be utilized by policy makers to achieve social equity in putting costs of ecosystem service losses on those responsible and using fees paid for lost services to restore those ecosystem services and thereby preserve them for the general public trust (National Research Council 2010).

2 Description of Ecosystem Services

Economic valuation of the ecosystem services provided by shellfish enhancement must start by identifying and describing those services. Based on four broad categories of ecosystem services developed by The Millennium Ecosystem Assessment (2005), Brumbaugh and Toropova (2008) organized the ecosystem services relevant to shellfish restoration or maintaining historical natural shellfish beds (Table 1). The literature describing the biological, physical and social processes underlying the provision of these ecosystem services is summarized below. It is important to note that the study of the environmental effects of shellfish enhancement is a large and rapidly expanding field, and the articles cited here represent only a small sample of the available literature. For more in-depth literature reviews the reader is referred to the comprehensive articles by Fisher and Mueller (2009), the National Research Council (2010) and Coen et al. (2011) and the regional reviews by Dumbauld et al. (2009) and Rice (2008). Furthermore, the current list of ecosystem services provided by shellfish may be incomplete—other benefits of shellfish doubtless exist without due recognition (Peterson and Lipcius 2003).

Table 1 Ecosystem Services Provided by Shellfish Enhancement

Provisioning	Commercial, recreational and subsistence fisheries Aquaculture Fertilizer and building materials (lime) Jewelry and other decoration (shells)
Regulating	Water quality maintenance Protection of coastlines from storm surges and waves Reduction of marsh shoreline erosion Stabilization of submerged land by trapping sediments
Supporting	Cycling of nutrients Nursery habitats
Cultural	Tourism and recreation Symbolic of coastal heritage

Source: Adapted from Brumbaugh and Toropova (2008)

2.1 Provisioning Services

Provisioning services are the products or goods people obtain from a restored or maintained shellfish population. They include foodstuffs and raw materials for building and manufacturing. The provisioning service of commercial shellfish fisheries and aquaculture production is often the predominant objective of enhancement programs. Shellfish are bought and consumed for their nutritional benefits as well as their taste—they are healthy sources of protein, rich in vitamins and minerals, low in fat and a good source of omega-3 fatty acids (Canadian Aquaculture Industry Alliance 2008). Despite substantial declines in U.S. shellfish landings,[5] the value of the shellfish industry to regional economies remains significant. Annual dockside value of oysters reached highs of more than $90 million per year in the 1990s (NOAA Fisheries Office of Habitat Conservation undated-a). Moreover, the full economic value of shellfish harvests goes well beyond dockside value; in addition to primary sales of the raw, unshucked product, there are economic benefits from secondary products

[5] U.S. landings of all oyster species in 1880 totaled around 154 million lbs. By the end of the 20th century, harvests had declined to just under 31 million lbs (King and McGraw 2004).

and services (e.g., shucking and packing houses, transport, manufacture of prepared oyster products and retail sales) (NOAA Fisheries Office of Habitat Conservation undated-a). In some areas shellfish fisheries and cultivating operations also contribute to public revenues through licensing and lease fees.

Some consumers choose to gather the mollusks themselves. Shellfish support major recreational fisheries in which people derive pleasure from an outdoor outing that offers them the opportunity to prepare and eat their own "catch." For example, shellfishing is a popular activity in Washington State, primarily along the Pacific Coast and the shoreline of Puget Sound. Thousands of local residents and visitors enjoy digging shellfish at local beaches and shoreline environments (U.S. Environmental Protection Agency 2006). It is estimated that the recreational razor clam fishery alone generates on average about 250,000 digger trips per year in southwest Washington counties. This recreational fishing effort represents about a $12 million influx of tourist and resident spending (on motels, food, gasoline, souvenirs, etc.) in coastal communities during winter and spring when business is otherwise slow (ORHAB Partnership 2002).

For thousands of years, the coastal indigenous peoples of North America have relied on the sea for most of their needs, and today shellfish from tidal flats remain an essential subsistence food source for Tribes and First Nations (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002). In the Pacific Northwest the relative ease with which large amounts could be harvested, cured, and stored for later consumption made shellfish an important source of nutrition—second only in importance to salmon (U.S. Environmental Protection Agency 2006). For example, Donatuto (2003) states that shellfish in the tidelands adjacent to the Reservation of the Swinomish Indian Tribal Community represent a vital subsistence and commercial resource for the Tribe, as well as an important point of cultural association for the Tribe's identity. Shellfish are employed in cultural ceremonies, incorporated in the common diet, and sold to support families on the Reservation. Similarly, in the Quinault language, the words ta'aWshi xa'iits'os mean "clam hungry", indicating the traditional dependence of this Tribe on shellfish as a subsistence food (ORHAB Partnership 2002).[6]

In addition to being harvested in commercial, recreational and subsistence fisheries for their edible meat, clams, oysters and other mollusks are removed for other purposes such as chicken grit (ground-up shells fed to chickens to help their gizzards digest food and to provide calcium for egg shells), natural water filters and buffers, calcium carbonate food supplements, and even as mulch for growing lavender plants (NOAA Restoration Center undated). Shucked oyster shell is also an ideal material with which to protect shorelines because the shell becomes tightly packed and is lighter than traditional shoreline protection materials (i.e., limestone rock). Furthermore, in some areas oyster larvae quickly recruit to the created reefs, indicating that reef maintenance is not likely to be a problem (Piazza et al. 2005).[7]

Pearls are by far the best known decorative product obtained from shellfish. Though not as hard or durable as mineral gems, pearls are nonetheless valued for their luster and scarcity; moreover, the naturally formed softness of pearls is part of their appeal, and has earned them the name "queenly gem" (Hisada and Fukuhara 1999). Natural and cultured pearls are manufactured into earrings/studs, necklaces, pendants, bracelets, rings and other jewelry. In 1997, the estimated world production value of fresh and salt water pearls was $472 million (this figure does not include China's production value) (Hisada and Fukuhara 1999). Also used for all kinds of decorative purposes is the inner shell

[6] Other Native American words also suggest the traditional importance of shellfish. For example, Paolisso and Dery (2008) note that the name "Chesapeake" is derived from Algonquin word "Chesepioc" (or Tschiswapeki), which translates into English as "great shellfish bay."

[7] While most oyster-producing states lack laws requiring oyster shells to be returned to the water, Washington is an exception. Washington regulations require that all oysters taken by sport harvesters on public tidelands be shucked (opened) on the beach, and the shells left on the same tideland and tide height where they were taken.

layer of oysters, which is called nacre or mother of pearl. Most jewelry made from shellfish is bought and sold in global markets. However, North American Tribes and First Nations still use the shells of mussels, clams, abalone and oysters to decorate woodcarvings and ceremonial apparel, as they did thousands of years ago (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002).

2.2 Regulating Services

Regulating services provided by shellfish enhancement are the benefits people obtain from the regulation of ecosystem processes. They are derived from the capability of shellfish to improve water quality through filtration, reduce shoreline erosion, and stabilize estuarine sediments.

2.2.1 Water Quality Maintenance

The water quality maintenance services of shellfish is a direct result of their suspension-feeding activity, which serves to reduce concentrations of microscopic algae (phytoplankton) and suspended inorganic particles in surrounding waters (Brumbaugh et al. 2006). In some coastal systems shellfish, through their feeding activity and resultant deposition of organic material onto the bottom sediments, are abundant enough to influence or control the overall abundance of phytoplankton growing in the overlying waters.[8] This control is accomplished both by direct removal of suspended material and by controlling the rate that nutrients are exchanged between the sediments and overlying waters (Ulanowicz and Tuttle 1992; Newell 2004). Encrusting, or fouling, organisms (e.g., tunicates, bryozoans, sponges, barnacles, and some polychaete worms) on oyster reefs and in the surrounding benthos are also suspension feeders and contribute to the overall filtering capacity of a reef (NOAA Fisheries Office of Habitat Conservation undated-a).

Due to this filtering of estuarine waters through feeding activity, shellfish can enhance water clarity and allow sufficient light penetration to support expansion of seagrass habitat (Newell and Koch 2004), an important estuarine “nursery area,” where juvenile invertebrates and fish are protected from predators. Ecosystem modeling suggests that restoring shellfish populations to even a modest fraction of their historic abundance could improve water quality and aid in the recovery of seagrasses, which can further reduce sediment resuspension and improve light conditions (Newell and Koch 2004). In addition, researchers suggest that robust populations of shellfish can suppress harmful blooms of phytoplankton, such as the “brown tides” that have occurred along the mid-Atlantic coast (Cerrato et al. 2004), and help modulate blooms of other types of harmful plankton, including “red tides” (Peabody and Griffin 2008).

As suspension feeders, shellfish also enhance water quality by concentrated deposition of feces and pseudofeces (particles collected on its gills that the oyster does not use as food) (Newell 2004; Newell et al. 2005). Increased biodeposition of organic matter in sediments leads to increased bacterial denitrification that can help to remove nitrogen from estuarine systems. The associated bacteria in sediments of an oyster bed can remove 20 percent or more of the nitrogen in oyster wastes, using the same process that is used in modern wastewater treatment plants (Shumway et al. 2003). Further,

[8] Field observations demonstrate that large populations of shellfish are capable of consuming a considerable fraction of the phytoplankton from overlying waters. For example, Haamer, and Rodhe (2000) showed that water passing a mussel bed in the sound between Sweden and Denmark was depleted of phytoplankton in the entire height of the fully mixed water column over a several kilometer long more or less continuous mussel bed. Similarly, field measurements by Grizzle et al. (2006) reveal a considerable fraction of suspended particulate matter is removed—up to 62 percent for the bivalve species *Mercenaria*—from waters overlying dense shellfish assemblages.

filter-feeding shellfish not only remove nitrogen from the water column; they also incorporate a high proportion of it into their tissues. Shellfish are approximately 1.4 percent nitrogen and 0.14 percent phosphate by weight. When the shellfish are harvested, the nitrogen is removed from the system, thereby recycling nutrients from sea to land (Shumway et al. 2003). Much of this nitrogen is in the relatively large shell and so when species with lighter shells, such as blue mussels, are harvested, less nitrogen will be removed (Newell 2004).

By mediating water column phytoplankton dynamics and denitrification, shellfish are likely to reduce excess nutrients that stimulate excessive plant growth in coastal waters, which often leads to low dissolved oxygen levels (hypoxia) as the phytoplankton die, a serious environmental problem in many aquatic ecosystems worldwide (Atlantic States Marine Fisheries Commission 2007). For example, 65 percent of U.S. coastal rivers, bays are moderately to severely degraded by nutrient pollution from agricultural practices, urban runoff, septic systems, sewage discharges and eroding streambanks (Bricker et al. 2008).

Shellfish filtration rates are a function of several environmental factors, including temperature, salinity, and suspended particulate concentration (Cerco and Noel 2005).[9] In addition, different shellfish species exhibit significant variation in filtration rates, with the rates generally increasing with species size (Powell et al. 1992). For large species, such as the eastern or American oyster, *Crassostrea virginica,* filtration rates have been estimated at 163 liters per gram (of oyster tissue) per day (NOAA Fisheries Office of Habitat Conservation undated-a). Newell (1988) calculated that the abundance of this species in Chesapeake Bay before 1870 was high enough that oysters could filter the entire volume of the bay in about three days (but after nearly a century of exploitation and habitat destruction, the reduced populations require 325 days to perform the same activity).

2.2.2 Protection of Shorelines and Sediment Stabilization

The protective influence of intertidal shellfish habitats on adjacent shorelines has been widely noted in the literature (Meyer et al. 1997; Piazza et al. 2005; Atlantic States Marine Fisheries Commission 2007; Borsjea et al. 2011). In some locations, oyster reefs and, likely, mussel beds that fringe the estuarine shoreline serve as natural breakwaters that protect the shoreline against the erosive force of wind- and boat-generated waves, thereby reducing bank erosion, protecting fringing salt marsh and decreasing loss of aquatic vegetation beds, such as eelgrass, behind the reefs.

Grabowski and Peterson (2007) note that oyster reefs are a living breakwater; consequently, they can rise at rates far in excess of any predicted sea-level rise rate. Moreover, a living, ecologically functional oyster reef can provide a more aesthetically pleasing and ecologically sound solution to coastal erosion problems than groins, breakwaters, sea walls or jetties (Marsh et al. 2002).

Oyster reefs that fringe the shoreline also tend to stabilize sediment (Peabody and Griffin 2008). The reduction in suspended sediment in adjacent waters improves water clarity, thereby possibly providing better opportunities for establishment of seagrasses and other species. Using a predictive model of Chesapeake Bay, Cerco and Noel (2007) assessed the impact of a tenfold increase in oyster biomass on three spatial scales, and suggested that the enhanced abundance of submerged aquatic vegetation is the most significant improvement to be attained from the water clarity benefits of oyster restoration.

It is also important to note that an oyster or mussel reef protruding only several centimeters above the bottom can help delaminate water flow across the bottom and assist in water column mixing processes (Atlantic States Marine Fisheries Commission 2007). This mixing and the formation of

[9] Filtration activity is defined in terms of clearance rate, i.e., the volume of water cleared of particles per time unit.

eddies around the reef structure functions to further reduce development of hypoxic conditions (NY/NJ Baykeeper undated). In addition, by disrupting flow on open bottoms or within tidal channels, reefs create depositional zones, usually downstream of the reef structure, that accumulate sediment and organic material (Lenihan 1999). These changes in hydrodynamics and material transport directly influence recruitment, growth, and other biotic processes of the shellfish and other organisms (e.g., finfish) that live on the reef (Atlantic States Marine Fisheries Commission 2007).

2.2.3 Carbon Sequestration

One potential regulating service that was not listed by Brumbaugh and Toropova (2008) is the role shellfish may play in sequestering (storing for long periods of time) carbon in the calcium carbonate of their shells, thereby reducing concentration of carbon dioxide, a "greenhouse gas" that contributes to global climate change (Rodhouse and Roden 1987; Peterson and Lipcius 2003; Hickey 2008; Tang et al. 2011). Carbon is absorbed naturally from the ocean as the calcium carbonate shell of the shellfish grows. Shells incorporated deep into the sedimentary strata beneath the seafloor and shells buried in soils on land will remain intact indefinitely, allowing the shellfish to provide a long-lasting service of preventing the carbon from re-entering the atmosphere (National Research Council 2010). In other words, it has been suggested that shellfish enhancement through the production of shells, can act as a "carbon sink." However, Allison et al. (2011) point out that this suggestion is based on the shell carbon content without considering factors such as respiration or the calcification process. Calcification induces shifts in the seawater carbonate equilibrium to generate dissolved carbon dioxide, and therefore actually releases carbon dioxide into the water (Chauvaud et al. 2003). While this process provides local chemical buffering against growing ocean acidification caused by increasing concentrations of atmospheric carbon dioxide (National Research Council 2010), the result is that shellfish enhancement may on balance prove to be a "carbon source."

Coen et al. (2011) conclude that the issue of whether the carbonate-rich deposits of live and dead shellfish are net carbon sinks or sources needs to be addressed in greater detail before it can be resolved to everyone's satisfaction as we discuss carbon sequestration and reduction of ocean acidification as a potentially important ecosystem service of shellfish enhancement. Noting the complexity of carbon cycles in marine systems, Allison et al. (2011) conclude that one should not "expect piles of oyster shells to be eligible for carbon payments anytime soon." According to the authors, the most useful contributions of shellfish enhancement to climate change mitigation is its compatibility with the maintenance of coastal environments that have known large carbon sequestration capacity. These coastal environments include tidal salt marshes, mangroves and seagrass meadows (Laffoley and Grimsditch 2009). In many cases the carbon sink capacity of these environments to mitigate climate change is being depleted by human activities (Macreadie et al. 2011).

2.3 Supporting Services

While not providing direct services themselves, supporting services are necessary for the production of all other ecosystem services. By creating structurally complex shell habitat and performing a wide array of ecological functions, shellfish populations can substantially modify benthic and pelagic communities at different trophic levels and alter energy flow and nutrient cycling over the scale of entire coastal ecosystems (Cranford et al. 2007).

2.3.1 Cycling of Nutrients

Because they are filter feeders, shellfish can greatly influence nutrient cycling in estuarine systems and maintain the stability of the ecosystem (NOAA Fisheries Office of Habitat Conservation undated-a). As noted earlier, oysters filter large amounts of phytoplankton and detritus (small organic particles) from the water column. As grazers of phytoplankton and other particles, these filter feeders couple, or join, the oyster reef to the water column. Some of the organic components resulting from shellfish metabolism serve as a nutrient source for benthic infauna, some enters the microbial loop, and some re-enters the water column. This flux or cycling of carbon, nitrogen and other essential materials is vital for the continuity and stability of any living system and acts to keep the system in balance (Peabody and Griffin 2008; NOAA Fisheries Office of Habitat Conservation undated-a).

In addition, Norkko and Shumway (2011) state that the burrowing, feeding and other activities of shellfish in the sediment can result in high levels of bioturbation and bioirrigation (i.e., mixing and flushing of sediments). These processes increase the transport of particulates, nutrients and oxygen across the sediment-water interface, and increase oxygen penetration into the sediment, thereby producing complex and favorable microhabitats that facilitate benthic communities with higher overall diversity.

2.3.2 Nursery Habitats

In addition to playing an important nutrient cycling role, some species of bivalve shellfish such as oysters and mussels form complex structures that provide refuge or hard substrate for other species of marine plants and animals to colonize, thereby enhancing biodiversity (Brumbaugh et al. 2006; NOAA Fisheries Office of Habitat Conservation undated-a). Lenihan and Grabowski (1998) note that these structures represent a temperate analog to coral reefs that occur in more tropical environments. Both kinds of structures are “biogenic”, being formed by the accumulation of colonial animals, and both provide complex physical structure and surface area used by scores of other species as a temporary or permanent habitat (Brumbaugh et al. 2006).

Older, maturing oyster reefs become larger and more complex, and provide greater habitat diversity. The extensive irregular surfaces of a reef provide 50 times the surface area of a similar sized flat bottom. These crevices provide good nursery habitat for a wide diversity of vertebrate and invertebrate organisms—worms, snails, sea squirts, sponges, crabs, and fish (Henderson and O’Neil 2003). By overcoming a survival bottleneck in the early life history of many fish and invertebrate species, oyster reefs enhance recruitment in those species, while other research indicates that oyster reef habitat contributes to fish and invertebrate production by providing refuge from predation and access to reef-associated prey resources (Peterson et al. 2003). The overall ecological result is greatly enhanced biodiversity in shellfish habitat compared to surrounding areas of the seabed (Atlantic States Marine Fisheries Commission 2007).

Perhaps most important from an ecosystem service perspective, certain types of shellfish offer the unique service of creating important habitats for other commercially or recreationally important species, particularly when they occur at high densities (Atlantic States Marine Fisheries Commission 2007). In fact, an oyster reef is the only habitat type that is itself a commercial edible species and a refuge and food for other marketable species (NOAA Fisheries Office of Habitat Conservation undated-a).[10] Among the commercial species of fish and invertebrates that use Atlantic and Gulf Coast oyster reefs at some time in their life cycles or prey upon oysters and associated fauna are flounder,

[10] The habitats created by shellfish can be classified into three major types: (1) reefs (veneer of living and dead animals), (2) aggregations (living and dead) and (3) shell (dead) accumulations (often called ‘shell hash’) (Atlantic States Marine Fisheries Commission 2007).

menhaden, herring, anchovies, spadefish, striped bass, cobia, croaker, silver perch, spot, speckled trout, Spanish mackerel, pinfish, butter fish, harvest fish, blue crab, stone crab, penaeid shrimp, black drum, and several species of mullet (Atlantic States Marine Fisheries Commission 2007; NOAA Fisheries Office of Habitat Conservation undated-a). On the Pacific coast, shells of the Pacific oyster placed at high density in the intertidal zone may provide excellent habitat for newly recruited Dungeness crab (Ruesink et al. 2005).

The potential size of the structure created by shellfish depends on the species. Available evidence suggests that reefs created by the Pacific oyster, *C. gigas* (mostly in the intertidal) and suminoe or Asian oyster *C. ariakensis* (mostly subtidal) are much smaller in size, occupy less area in estuaries, and are a more heterogenous mix of shell and sediment compared with eastern oyster reefs (Ruesink et al. 2005). The Olympia or West Coast oyster, *Ostreola conchaphila*, is not a reef builder, but instead is usually found attached to rocks or dead shells (Peabody and Griffin 2008). Nonetheless, Olympia oyster aggregations (beds) have high biodiversity because they provide a physical habitat structure ideal for juvenile fish and crustaceans, worms, and foraging nekton and birds (Peabody and Griffin 2008). Shellfish habitat—whether it is a living assemblage or an accumulation of dead shells—provides hard substrate for the attachment of many species that would not be present in areas consisting only, or mainly, of soft sediments. The overall ecological result is greatly enhanced biodiversity in shellfish habitat compared to surrounding areas of the seabed (Atlantic States Marine Fisheries Commission 2007).

The structures used in shellfish aquaculture (racks, cages, nets, ropes, trays and lines) also provide habitat by providing surfaces for attachment of other organisms (Shumway et al. 2003). For example, macroalgae and epifauna growing upwards from protective plastic mesh used in bottom clam culture can substitute for natural seagrass habitat as a nursery area for mobile invertebrates and juvenile fish (although this nursery habitat is removed and cleaned at harvest) (Coen et al. 2007). In comparison to unplanted adjacent sandflat, the epibiotic habitat growing on aquaculture bottom netting had a 42 (fenced lease) to 46 (open lease)-fold enhancement of mobile invertebrates and a 3 (fenced lease) to 7 (open lease)-fold enhancement of juvenile fishes (Coen et al. 2007). The subtidal rack and bag systems used to rear oysters in Southern New England and parts of the Northeast can act as refugia for a variety of marine organisms, including the juvenile stages of various species of commercially valuable finfish (Rice 2008).

The influences of shellfish habitat on associated populations, assemblages and ecological processes can extend beyond the shellfish reefs and beds into adjacent habitats. For example, shellfish reefs and beds diversify the seascape to enhance the synergistic benefits of multiple habitat types, such as creating corridors between shelter and foraging grounds (Peterson and Lipcius 2003). Furthermore, shellfish reefs and beds can support the creation of other habitat types. As discussed previously, oyster reefs can dampen waves and thus reduce the erosion of salt marsh faces and help stabilize submerged aquatic vegetation beds; they can filter estuarine waters, thereby enhancing their clarity and allowing sufficient light penetration to support expansion of seagrass habitat; and they can increase the nutrients available to seagrasses through the deposition of organic matter and waste by-products (Peterson and Heck Jr. 1999). Oysters probably generate greater per capita organic deposition than other bivalve types because of their high filtration rate and capacity to discharge pseudofeces and thereby continue filtration under conditions of high turbidity (National Research Council 2010). Thus, oyster reefs can support at least two other important habitat types within an estuary (NY/NJ Baykeeper undated). Seagrasses and salt marshes constitute additional key habitats in estuaries and provide food, habitat and nursery areas for many species (NOAA Fisheries Office of Habitat Conservation undated-a). These improvements to the ecological integrity of other habitats may ultimately lead to additional increases in the production of finfish and invertebrates targeted in commercial and recreational fisheries (National Research Council 2010).

2.4 Cultural Services

Cultural services provided by shellfish are the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences.

As discussed above, some shellfish support recreational fisheries which offer participants the opportunity to gather edible shellfish themselves. People who engage in this pastime typically do so because they derive aesthetic and social benefits from the experience as well as sustenance. In addition, shellfish can play an important role in recreation and tourism by creating habitat for fish, allowing recreational fisher use, and can improve adjacent beach water quality, resulting in more desirable areas for tourists and local residents to visit (Henderson and O'Neil 2003). As an example of the sociocultural importance of recreational shellfishing, Bauer (2006) noted that razor clam harvesting, cleaning, cooking, eating and canning have been an important focus of family relationships and local culture in Washington coastal communities for many generations. To underscore this importance Bauer provides a quote from Dan L. Ayres, Coastal Shellfish Lead Biologist, Washington Department of Fish and Wildlife: "The cultural significance of joining with friends or family to successfully brave the natural elements to take home a sport limit of fresh razor clams cannot be understated."

Community support and involvement in shellfish enhancement projects can heighten public awareness of the need to rehabilitate and conserve marine and estuarine ecosystems. For example, Reynolds and Goldsborough (2008) argue that hands-on oyster reef restoration projects are not only complimentary to, but a critical component of, advancing a broader environmental policy agenda aimed at reducing nutrient loading in coastal waters. Public involvement and ownership of shellfish restoration projects is expected to ensure continuation of the projects into the future and the ability to address additional environmental issues as these arise.

In recent years, several U.S. coastal communities have become involved in restoring shellfish habitats in their areas, often in partnership with state and federal agencies and private, non-profit organizations. *Promoting Oyster Restoration Through Schools* is a community-based restoration and educational program focusing on the importance of oyster populations in the Delaware Bay ecosystem. The program utilizes the oyster as a vehicle to acquaint school children, grades K-12, with the Delaware Estuary and basic scientific concepts (Haskin Shellfish Research Laboratory 2008).

The *South Carolina Oyster Restoration and Enhancement Program* has developed numerous education tools to convey the importance of oysters to its volunteers, including online tutorials, presentations, laboratory exercises, simple experiments, observational field studies and volunteer-friendly sampling and monitoring techniques to educate and involve individuals and groups of all ages (Hodges et al. 2008). The program's underlying premise is to empower citizens to take responsibility for water quality and to encourage them to acquire a "vested interest" in water resources through investment of personal time and energy (Hadley et al. 2008).

A project to restore about one acre of oyster reef in the West Branch of the Elizabeth River, Virginia, provided an opportunity for local middle-school students to become actively engaged in a resource restoration project in their home waters (NOAA Chesapeake Bay Office 2008). The project stimulated public awareness of the ecological value of oyster reefs while instilling a sense of community stewardship in local restoration work. According to the NOAA Chesapeake Bay Office (2008), the project succeeded because of the development of a partnership approach to oyster restoration between a local community, a conservation organization, a state agency and a federal agency.

Mass et al. (2008) note that one of the major successes of the eastern oyster restoration project in the Bronx River has been the involvement of multiple Bronx River community groups. Various groups in the Bronx have helped construct the pilot reefs, monitored for spat settlement, maintained oyster "gardens" in the area, and educated the public about the benefit of oyster reefs and their importance to the ecosystem. A community-based project to transplant blue mussels into a degraded tidal salt pond in Portsmouth, New Hampshire, garnered city involvement and increased local awareness of the pond as an ecosystem rather than a sewage lagoon (McDermott et al. 2008).

According to Burke and Menzies (2010), an indication of the value of shellfish resources to the residents of Drayton Harbor, Washington is the volunteer contributions of time and money these residents made to improve water quality in the area and restore a community oyster farm. Between 1998 and 2008, over 38,000 volunteer hours were spent on this shellfish restoration project. The authors report that this public investment in volunteerism has contributed to the social capital necessary for effective local government in the area.

In addition to these community-based shellfish restoration efforts, thousands of people participate each year in shellfish celebrations that raise money for non-profits, promote community engagement in local environmental issues and segue nicely with the growing interest in local food and connecting people with local foods and traditions (U.S. Environmental Protection Agency 2006). A good example of such a celebration is the Annual West Coast Oyster Shucking Championship and Washington State Seafood Festival held in Shelton, Washington. "OysterFest" is the primary fund raising activity of the Shelton SKOOKUM Rotary Club Foundation, which uses the earnings to support a broad array of community organizations and events (Shelton SKOOKUM Rotary Club Foundation 2008).

Shellfish fisheries and aquaculture can also indirectly bring local environmental problems to the attention of nearby communities. U.S. public health standards under which shellfish fisheries and aquaculture operate demand clean waters and commercial shellfish harvest can only take place in waters that have been certified under the National Shellfish Sanitation Program (Shumway et al. 2003). The standards of this program fostered the first estuarine/marine monitoring programs, and are the most stringent of all U.S. water quality classifications, far exceeding those required for swimming. As a result, the presence of shellfish fisheries and aquaculture often results in increased monitoring of environmental conditions of estuaries and coastal waters. Moreover, the economic hardships suffered by communities following closure of shellfish fisheries and culture operations due to water contamination have often provided the political impetus for improvement in sewage treatment plants or programs to fix local septic systems (Shumway et al. 2003).

Public participation in shellfish restoration projects can also foster an appreciation of cultural heritage. In coastal communities throughout the U.S., shellfish are cultural icons, reflecting traditions and a way of life dating back generations (Brumbaugh et al. 2006). For example, the "watermen" in Chesapeake Bay area communities who make a living by harvesting oysters derive more than material gain from their work—for them it is entire lifestyle. What's more, there is a growing recognition among the broader public that the livelihood of these individuals is an integral part of what is worth preserving in America's coastal areas (Wasserman and Womersley undated). Similarly, shellfish aquaculture is part of the cultural history in parts of Europe. Shellfish culture in Europe has become a family tradition, where the rights and also the skills stayed within families (Wijsman 2008).

As discussed in Section 2.1, the harvesting of shellfish for food and cultural purposes is also a longstanding practice deeply rooted in some aboriginal communities in North America and elsewhere (Kingzett and Salmon 2002). For instance, the Wampanoag Tribe, Massachusetts' only federally recognized Native American Tribe, is investing time and money into culturing bay scallops, as bay scallops historically played an important cultural role for the Tribe (Trauner 2004). Such tribal projects can help preserve traditional ecological knowledge as well as provide an important source of cash income (Harper 2008).

Western Washington Tribes continue to use shellfish for subsistence, economic and ceremonial purposes (U.S. Environmental Protection Agency 2006), and the Tribes are closely involved in efforts to rebuild stocks of Puget Sound's native oysters (Peter-Contesse and Peabody 2005; Kay 2008). Moreover, shellfish grounds have become important to some Washington Tribes by affirming treaty rights that entitle them to fish and hunt for subsistence and commerce on traditional lands and water (Barry 2008). Federal courts have upheld the Tribes' treaty rights to harvest shellfish in Puget Sound, including on private tidelands under certain circumstances.[11] The exercise of these treaty rights helps Tribes preserve their self-sufficiency and cultural autonomy (ORHAB Partnership 2002).

2.5 Ecological Uncertainty

Freeman (1999) states that the first steps in economic valuation of ecosystem services are to determine the size of the environmental change affecting ecosystem structure and function and determine how these changes affect the quantities and qualities of ecosystem service flows to people. However, it is important to recognize that many of the reported changes in ecosystem structure and function associated with shellfish enhancement are a source of scientific debate. Major gaps in knowledge include how native and introduced shellfish species influence nutrient cycling, hydrodynamics, and sediment budgets; whether other native species use them as habitat and food; and the spatial and temporal extent of direct and indirect ecological effects within communities and ecosystems (Ruesink et al. 2005).

For example, one of the reported ecosystem services provided by restoration of oyster reefs and other bivalve-dominated habitats, the grazing of phytoplankton populations, was the focus of recent articles (Pomeroy et al. 2006; Pomeroy et al. 2007), which concluded that filtration by the eastern oyster in Chesapeake Bay, either at historical densities or at current restoration target densities, is insufficient to reduce the severity of phytoplankton blooms and resulting hypoxia in the Bay because of temporal and spatial mismatches between eastern oyster filtration and phytoplankton (see reply by Newell et al. (2007)). In a literature survey following this debate, Coen et al. (2007) found that several researchers expressly noted that the system-level effects of oyster filtration have been poorly quantified, especially as they might relate to any specific native oyster reef restoration project. The authors noted that the potential benefits of filtration by oysters as stated in the popular press ignore the realities of the scale of restoration required to achieve such benefits. More recently, a debate arose in the literature over the potential of mussel farming as a nutrient reduction measure in the Baltic Sea (Stadmark and Conley 2011; Rose et al. 2012; Stadmark and Conley 2012).[12] The issue of shellfish and carbon sequestration brings up yet another area of controversy (see Section 2.2.3).

Peterson and Lipcius (2003) suggest that planners and decision makers would be able to make more informed decisions about future restoration actions if they were armed with explicit estimates of the probabilities of a suite of alternative outcomes associated with each restoration alternative. However, due to the interconnectedness of the various elements of an ecosystem and the variety and

[11] Although the Tribes have no claim to the oysters, native or otherwise, that are planted on privately owned or leased tidelands, they do have a right to 50 percent of the shellfish that existed prior to any stock enhancement efforts. An agreement was signed between the 13 treaty Tribes of Puget Sound and the commercial shellfish growers in Washington State. Under this agreement the Tribes have agreed not to harvest from qualifying shellfish beds from qualified commercial growers (Barth 2008).

[12] Rose et al. (2012) argue that the considerable uncertainties surrounding the application of shellfish enhancement, "'nutrient bioextraction'' strategies to nutrient management are attributable in part to the limited number of direct field trials and also to the inherent variability across coastal ecosystems worldwide. The authors conclude that a more thorough evaluation of nutrient assimilation and regeneration resulting from shellfish biodeposition within an ecosystem context is needed to reach informed conclusions for each location under consideration.

complexity of ecological outputs (National Research Council 2004), it is extremely difficult to identify the probability that an impact on the biological and human environment will occur and to weigh this probability against the magnitude of the possible impacts (Evans 1979). The ability of economists to place economic valuations on ecosystem services is contingent on a concerted effort to measure and document these services in the field. Consequently, ecological uncertainty propagates through to uncertainty about economic outcomes (Dorrough et al. 2008).

In order to start to address uncertainties about the ecological effects of shellfish enhancement projects emphasis has been placed on project monitoring, i.e., systematic data collection that indicates progress toward identified criteria, performance standards and ecological goals (NOAA Fisheries Office of Habitat Conservation undated-b). Peterson and Lipcius (2003) argue that monitoring the performance of the restoration and then adaptively modifying the scale or even type of restoration in response to documented performance of the restoration could legitimately be included among the costs of restoration. In 2004, participants at a Sea Grant sponsored workshop proposed a set of sampling criteria and methodologies to provide standardized population and ecosystem measures for assessing the success of oyster restoration projects (Coen et al. 2004). In addition, The Nature Conservancy and partners have developed a nationwide network of shellfish restoration sites where quantitative approaches are used to monitor ecosystem services and outcomes associated with restoration projects (Brumbaugh and Toropova 2008).

In recent years, modeling has also been increasingly used as a means to acquire a better understanding of the impacts of shellfish enhancement on water quality, nutrient cycling and benthic processes. The application of mathematical models provide a means of quantitatively evaluating ecosystem goods and services which result from shellfish farming in coastal environments. For example, nutrient enrichment interactions with shellfish aquaculture only recently began to be assessed through modeling efforts, mostly within the past decade (Burkholder and Shumway 2011). One such modeling effort was the use by Landeck-Miller and Wands (2009) of a mechanistic numerical model of Long Island Sound eutrophication processes, the System Wide Eutrophication Model, to assess the potential of shellfish aquaculture as an additional means of removing nitrogen from Long Island Sound. For reviews of modeling studies pertaining to the environmental impacts of shellfish aquaculture see Burkholder and Shumway (2011) and Grant and Filgueira (2011).

3 Economic Valuation of Ecosystem Services

The economic concept of value has been broadly defined as any net change in human well-being or welfare. In economic analysis, any action which increases welfare is a benefit and any action which decreases welfare is a cost. In assessing the value of a project to protect or restore a shellfish population, the economist is interested in estimating how much the welfare of one person or society at large would change as a result of that project.

Economic welfare includes what economists call consumer surplus and producer surplus. Consumer surplus is the net value consumers receive from a good or service over and above what they actually pay for the good or service. Producer surplus (also called economic rent) is the difference between what producers actually receive when selling a product and the amount they would be willing to accept for the product. While not an exact measure of social welfare, the sum of the consumer and producer surplus that results from the ecosystem services provided by a shellfish enhancement project provides a useful approximation of the project's net benefits.

Converting the benefits of ecosystem services to a common comparable unit (dollars) so as to sum them often represents a major challenge to economics (Peterson and Lipcius 2003). While economists possess an array of methods to assess consumer and producer surplus in monetary terms, no single method can capture the total value of the many, disparate ecosystem services provided by a complex natural asset such as an oyster reef (Johnston et al. 2002). Moreover, although the value of some services can be readily monetized, the value of others can be done so only with great difficulty and uncertainty (Johnston et al. 2002). For example, estimation of consumer surplus is relatively straightforward if services are traded in traditional markets with market prices and values (e.g., commercial shellfish, shell for road building). However, as discussed in Section 1.1, many of the benefits of the ecosystem services provided by shellfish enhancement (e.g. water filtering and enhanced water quality, habitat for other organisms) accrue directly to people without passing through the market economy.

The first section of this chapter provides an overview of methods to estimate dollar measures of the value of ecosystem services. Each economic valuation method has strengths and weaknesses, and each service has an appropriate set of valuation methods. Furthermore, some services may require that several methods be used jointly (Farber et al. 2002; Carson and Bergstrom 2003). The second section demonstrates how economic valuation methods can be used to place a value on the services provided by shellfish enhancement. Where possible, examples drawn from the literature are used to illustrate applications of these methods.

3.1 Methods

There are two general types of approaches for estimating economic welfare gains (or losses). The first approach, which is to conduct primary research, can be subdivided into indirect (revealed preference) and direct (stated preference) methods. This approach requires the collection of new data, which may be costly and time-consuming (Dumas et al. 2005). Consequently, some researchers have adopted the second approach, commonly called benefit transfer, whereby existing valuation information for an ecosystem service is used to estimate the value of a similar ecosystem service.

3.1.1 Primary Research Approach

3.1.1.1 Revealed Preference Methods

Revealed preference techniques estimate the value of an ecosystem service using market data and consumer characteristics, activities and purchases (Isaacs et al. 2004). The major strength of indirect approaches is that they are based on data reflecting actual market choices, where individuals bear the actual costs and benefits of their actions (Dumas et al. 2005). The most common revealed preference methods are the market price method, hedonic pricing method, travel cost method and cost-based methods.

The market price method uses the prices of goods and services that are bought and sold in commercial markets to determine the value of an ecosystem service (King and Mazzotta 2000a). By measuring the change in producer and consumer surplus after the application of a change in production or price, the value can be determined (Carson and Bergstrom 2003). The primary shortcoming of this method is that it only takes into account the market components of the value of ecosystem services.

The travel cost method estimates the number of recreational trips an average person takes to a specific site, as a function of the cost of travelling to that site, the comparative costs of travelling to substitute sites, and the quality of the recreational experience at the sites.[13] The basis of the method is the assumption that the recreational experience is enhanced by high quality sites (e.g., clean water, abundant recreational fisheries), hence the net willingness to pay for—and value of—recreational trips depends on site quality. Travel cost models require data on participation rates, cost of travel to sites and site quality (Johnston et al. 2002).

Cost-based methods, which include the damage cost avoided, replacement cost and substitute cost methods, are related methods that estimate values of ecosystem services based on either the costs of avoiding damages due to lost services, the cost of replacing ecosystem services, or the cost of providing substitute services (King and Mazzotta 2000c). These methods do not provide strict measures of the economic value of ecosystem services, but rather provide rough indicators of the value by assuming that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. For example, the replacement cost method might identify a project for providing the same services and calculate the cost of construction for that project (Carson and Bergstrom 2003).

King and Mazzotta (2000c) note that that the key assumption of cost-based methods may not necessarily be valid. Just because an ecosystem service is eliminated is no guarantee that the public would be willing to pay for the identified least cost alternative merely because it would supply the same benefit level as that service. Without evidence that the public would demand the alternative, cost-based methods are not valid estimators of ecosystem service value. King and Mazzotta conclude that cost-based methods are most appropriately applied in cases where damage avoidance or replacement expenditures have actually been, or will actually be, made.

[13] The random utility model is a variation of the travel cost method. Unlike the traditional travel cost model, which focuses on one recreation site, the random utility model uses information from multiple recreation sites. Individuals choose a recreation site based on differences in trip costs and site characteristics (e.g., water quality) between the alternative sites. Statistical analysis of the relationship between site characteristics and recreationists' site choices enables estimation of any consumer surplus changes arising from any changes in site characteristics (Dumas et al. 2005)

3.1.1.2 Stated Preference Methods

Stated preference methods involve questioning survey respondents to determine changes in consumer surplus. The contingent valuation and conjoint/choice analysis methods are examples of these methods. The contingent valuation directly elicits net willingness to pay as a measure of the value of a specified level of environmental quality. As an example, a survey might ask respondents' willingness to pay for water that is acceptable for swimming and other activities (Henderson and O'Neil 2003). The conjoint/choice survey format asks respondents to choose between bundles of environmental assets, which differ across their physical, biological, aesthetic and/or money dimensions (Johnston et al. 2002).

The major weakness of stated preference methods is their hypothetical nature. The results of the methods are often highly sensitive to what people believe they are being asked to value, as well as the context that is described in the survey.[14] In most cases, the survey is designed to include a description of a realistic or plausible scenario that is comprehensible to the respondent. For example, for previously highly polluted systems such as the Chesapeake and the Upper Narragansett, where beach closings due to pollution are common, the hypothetical nature of stated preference methods is not a limit—beach closures and swimming bans actually happened (Henderson and O'Neil 2003). In some surveys, however, respondents may be placed in unfamiliar situations in which complete information may not be available. At best, respondents give truthful answers that are limited only by their unfamiliarity. At worst, respondents give unconsidered answers due to the hypothetical nature of the scenario (Dumas et al. 2005).

Notwithstanding these problems, stated preference methods may be the only kind of valuation technique suitable in many circumstances. In particular, they are the only way of measuring non-use values, also referred to as passive-use values, which include such values as existence value and option value. Existence value is the welfare obtained from the knowledge that an environmental asset exists in a certain condition, without directly using it. Option value is the welfare obtained by retaining the option to use an environmental asset at some future date.

3.1.2 Benefit Transfer Approach

As noted above, the benefit transfer approach is used when limited time or funding preclude costly data collection and the development of new consumer surplus estimates. With benefit transfer, value estimates from one or more previously conducted valuation studies that used revealed or stated preference methods are spatially and/or temporally transferred to a new study (Dumas et al. 2005). The literature on benefit transfer generally describes four types of approaches: value (benefit estimate) transfer; value function transfer, meta regression analysis and preference calibration (Rosenberger and Loomis 2001; Dumas et al. 2005; Ready and Navrud 2005).

Value transfer uses summary measures of the environmental benefit estimates directly. (Dumas et al. 2005). The approach encompass the transfer of a single (point) benefit estimate from an existing study, or a measure of central tendency for several benefit estimates from a previous study or studies (such as an average value). The primary steps to performing a single point estimate transfer include identifying and quantifying the changes in, say, recreational use at a study site, and locating and

[14] Economists acknowledge that questions of validity, bias, and reliability persist in the use of the contingent valuation method and other survey-based methods to value the non-market components of ecosystem services. In 1992, the National Oceanic and Atmospheric Administration commissioned a blue ribbon panel to advise the agency on the use of the contingent valuation method (Arrow et al. 1993). The panel concluded that the contingent valuation method can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resource damages as long as certain sampling and survey design guidelines are followed.

transferring a "unit" consumer surplus measure (Rosenberger and Loomis 2001). Consequently, this approach is best suited for situations where the projected impacts of a project or policy can be measured in fairly homogeneous, divisible units (Ready and Navrud 2005).

Function transfers are a more rigorous approach whereby a benefit or demand function is transferred from another study. The benefit function statistically relates peoples' net willingness to pay to characteristics of the ecosystem and the people whose values were elicited (King and Mazzotta 2000b). When a benefit function is transferred, it can be adapted to fit the specific characteristics of the site of interest, thus allowing for more precision in transferring benefit estimates between contexts (King and Mazzotta 2000b; Rosenberger and Loomis 2001). Ready and Navrud (2005) note that value function transfer will work well only if a) there is sufficient variation at the existing study site in the attributes of the ecosystem, b) there is sufficient variation at the existing study site in the characteristics of the user population, c) the attributes of the ecosystem and the population at the site of interest fall within the range of the original data at the study site, and d) consumer preferences for the ecosystem services are similar at the study site and site of interest.

Meta regression analysis offers the advantage of combining results from several original valuation studies. With this method, benefit estimates gathered from multiple studies serve as the dependent variable in regression analysis, and characteristics of the individual studies serve as the independent variables. According to Dumas et al. (2005), meta regression analysis may be used to control for differences in functional form and other methodological differences across studies as well as differences between the study sites and site of interest. Problems with this method include reporting errors and omissions in the original studies; inconsistent definitions of environmental commodities and values; and large random errors (Dumas et al. 2005).

Smith et al. (1999) proposed the method of preference calibration as a solution to the problems associated with benefit function transfer and meta regression analysis. Rather than computing a unit value or constructing a statistical function describing how unit values change with economic or demographic variables, the same existing studies can calibrate a specific preference function. Dumas et al. (2005) note that a major benefit of preference calibration is its recognition that net willingness to pay is constrained by income in situations involving large changes in policy variables. However, the authors also list several problems with preference calibration: it does not tailor the benefit estimates to the demographics and other characteristics of the site of interest as does benefit function transfer and meta regression analysis; it is more time consuming than benefits function transfer due to the increased analytical burden; and it has yet to be vetted by tests of transfer accuracy.

Benefit transfer studies have been carried out at various geographical scales and for a wide range of ecosystem services. However, there is still no consensus over the accuracy and precision of the technique for a number of reasons, including disagreement over the principles that should underlie benefit transfer, over the required standards of accuracy for transfers, and over the best valuation methods for developing benefit transfer (Hanley et al. 2006). Furthermore, the number of original studies available is insufficient for many purposes. It is true that the international peer-reviewed literature in the field of economic valuation of ecosystem services has grown substantially in recent decades (Wilson and Farber 2008). In addition, access to this literature is facilitated by the availability of searchable online databases.[15] However, the lack of benefit studies across multiple contexts remains one of the significant challenges to the growth and sustainability of these online databases

[15] These online databases include the one maintained by the National Ocean Economics Program (2012), which is the first portal dedicated to compiling and organizing bibliographic information on valuation studies specific to coasts and oceans, and the Environmental Valuation Reference Inventory (2011), an international database of studies on the economic value of environmental benefits and human health effects that has been developed specifically as a tool for the benefit transfer approach.

(Wilson and Farber 2008).[16] For example, with particular reference to the nutrient cycling benefits of oyster reefs and other estuarine habitats, Piehler and Smyth (2011) note that transferability needs to exist within a system (e.g., are rates of nitrogen removal the same across habitat types and landscapes) and between systems (e.g., are habitat-specific nitrogen removal rates similar from estuary to estuary) before these processes can be modeled effectively at the ecosystem level and extended to economic evaluations.

3.2 Applications

This section reviews studies that have used economic valuation methods to place a value on the services provided by shellfish enhancement. A sample of these studies and their findings is provided in Table 2.

Table 2 Examples of Valuation of Ecosystem Services Provided by Shellfish Enhancement

Commercial fisheries and aquaculture	Lipton (2008c) projected that the net returns to harvesting oysters in Maryland and Virginia over a 10-year time horizon were $12.8 million.
	Lipton (2008a) estimated the annual gain in consumer surplus from an increase in Chesapeake Bay harvest of 2.57 million bushels of oysters to be $11.6 million.
Recreational fisheries	English (2008) reported an average per-trip value of $21.40 for recreational shellfishing in southeastern Massachusetts.
Water quality maintenance	Hicks (2004) determined that the water quality improvements resulting from restoring 1,890 acres at 73 reef sites in Chesapeake Bay would annually generate $640 thousand in benefits for recreational anglers.
	Newell et al. (2005) derived an annual replacement value of nitrogen removal by oyster reefs in the Choptank River, Maryland of $314,836, or $181 per hectare.
Cycling of nutrients and creation of habitat	Grabowski and Peterson (2007) estimated the value of enhanced commercial fish production by oyster reefs in the southeast United States to be $3,700 per hectare per year.
	Isaacs et al. (2004) found that the average annual net willingness to pay among resident saltwater recreational fishermen to maintain access to fishing over Louisiana's oyster reefs was $13.21.

3.2.1 Provisioning Services

As discussed in Section 2.1, shellfish can support important commercial, recreational and subsistence fisheries where their growth is abundant and the waters are suitable (Henderson and O'Neil 2003).

[16] Smith and Pattanayak (2002) describe one factor that has limited the number of economic valuation studies: "To be published, non-market valuation research generally must introduce a new method. Field journals in environmental economics are usually not interested in new estimates of the benefits from improving a given environmental resource for their own sake. Updating results for a specific application, such as the demand for sport-fishing recreation or new estimates of the marginal willingness to pay for improvements in air quality may have policy value but usually will not be considered important enough to occupy scarce journal space."

This section reviews studies that have placed a value on these provisioning services of shellfish enhancement.

3.2.1.1 Commercial Fisheries and Aquaculture Operation

Using the market price method, the value of commercial shellfish fisheries and aquaculture operations is comparatively easy to evaluate, as they generate products that are bought and sold in markets and, therefore, have observable prices. Welfare received by society from these activities is represented by the producer surplus that accrues to the harvesters, processors and retailers of the products and by the consumer surplus accruing to those who purchase and consume the harvested products.

Producer Surplus Derived from Commercial Shellfish Fishing or Culture

The appropriate measure of producer surplus for a shellfish fishing or aquaculture enterprise attempts to determine income or gross revenues net of all the firm's costs of production, including the costs of labor and capital. Gross revenues can be estimated based on the dockside price paid to the fisher/farmer for the shellfish and the quantity sold. Price and landings data can generally be obtained from government agencies or trade journals. Calculating expenses is typically more difficult due to a lack of publically-available cost data. Even with cost data an issue often arises related to the cost of a person's labor. Labor costs must consider the opportunity cost of labor, that is, what an individual could have earned if they had not spent that time shellfish fishing or farming (Lipton 2008b). It is conceivable that some individuals have greater earning potential when they are not on the water, but forgo higher income because they prefer working the water to other forms of employment (Lipton 2008b).

The production value of shellfish goes well beyond dockside value of the raw, unshucked product (King and McGraw 2004); the economic benefits of shellfish fishing or aquaculture to other market levels such as processors, wholesalers and retailers must also be considered.[17] Estimates of welfare measures for the wholesaler/processor segment of the industry must take into account that wholesale product prices double count the dockside value, that is, the price of a wholesale shellfish includes the price paid to the fisher/farmer for the shellfish plus the expense of adding value by processing, packaging and transporting the product, plus the profit to the processor (Lipton 2008b). Only that increase in profit to the processor is a potential welfare gain from shellfish enhancement, and even of that, only the profit that they earn from shellfish over and above they might earn from investing in processing some other product would count. For example, if the processors earn greater profits because they no longer have to transport shellfish from other regions, it would be that increased profit that would be the measure of the welfare gain, not the total value of their processing output. Even then, over time, market factors might shift to eliminate these benefits, and the welfare gains to local processors might lead to welfare losses to other processors outside the region (Lipton 2008b).

A number of studies have estimated, at least partially, the producer surplus generated by shellfish restoration projects, or collected industry information that would support such an analysis. Lipton (2008a) estimated the net returns to harvesting oysters in Maryland and Virginia over a 10-year time horizon using a harvesting cost estimate by Wieland (2006). The net present value of the stream of net revenues using a 2.6 percent rate of discount was $12.8 million. In an economic assessment of an oyster restoration project in Apalachicola Bay, Florida, Berrigan (1990) calculated the revenues from

[17] Some shellfish fishing or cultivating operations harvesters deliver directly to restaurants or other retail outlets, but it is more common for harvesters to sell shellfish either to wholesalers or to processors. Wholesalers repack shellstock into sacks, boxes, or bushels and sell them to processors or directly to restaurants or retailers. Processors produce raw shucked shellfish; prepared raw halfshell shellfish; and smoked, cooked, or canned shellfish (Muth et al. 2002).

shellfish harvest and revenues generated by added value through wholesale and retail sales. The British Columbia Ministry of Agriculture and Lands (2005) estimated the production economics for seeded sub-tidal geoduck grow-out production in British Columbia using a farm model. Coffen and Charles (1991) investigated the determinants of shellfish aquaculture production in Atlantic Canada through the estimation of Cobb-Douglas production functions, relating production output to several independent input variables. The statistical analysis was carried out for both mussel and oyster culture, based on data collected in a survey of aquaculturists. Muth et al. (2002) estimated the per unit processing costs of post-harvest treatment of Gulf-harvested oysters. The production costs of shellfish culture in Willapa Bay, Washington, were examined by Bonacker and Cheney (1988). Frank Harmon Architect and Olympus Aquaculture Consulting (2008) estimated the profitability of a hatchery in Virginia for the production of oyster larvae and spat. Adams et al. (1993) projected positive net returns for a hard shell clam culture operation in Cedar Key, Florida.

Lipton et al. (1992) note that the producer surplus generated from commercial shellfish fisheries will depend on how the shellfish resource is managed. Net benefits to producers will likely be less under an open access fishery management regime than if a bottom leasing program is instituted.[18] This expectation follows from the well-known result of "rent dissipation" in open access management regimes (Lipton et al. 1992).

In recent years, researchers have made greater use of modeling tools to improve the economic performance of shellfish aquaculture operations. For example, the Farm Aquaculture Resource Management (FARM) modeling framework applies a combination of physical and biogeochemical models, bivalve growth models and screening models for determining shellfish production (Longline Environment Ltd. 2012). The FARM model will eventually enable a detailed analysis of the production of a full range of cultivated shellfish species, including various species of mussels, oysters and clams. In a study of five different shellfish aquaculture systems, ranging from bays to lochs, where different species and aquaculture techniques are being used, Ferreira et al. (2007) added a Cobb–Douglas function to the FARM model to determine the optimal stocking density in terms of maximum economic profit. Silva et al. (2011) used the model in conjunction with other modeling tools to predict potential production, economic outputs and environmental effects at proposed shellfish aquaculture sites.

Consumer Surplus Derived from Commercial Shellfish Fishing or Culture

Consumer surplus will often be increased by shellfish enhancement because the seafood eating public will have available a quality and greater quantity of shellfish at a lower price. Lipton (2008a) and Lipton (2008c) approximated the consumer surplus benefit from a restored Chesapeake Bay oyster fishery using an estimated inverse demand curve. The author notes that this is a simple approach and it has limitations caused by failure to consider the entire system of demand and supply equations; the system of equations for each of the different types of oysters; a demand specification inconsistent with traditional economic theory; and extremely limited data (Lipton 2008a). The annual gain in consumer surplus from an increase in Chesapeake Bay harvest of 2.57 million bushels of oysters was estimated to be $11.6 million.

3.2.1.2 Recreational and Subsistence Fisheries

Comparatively few studies have assessed the recreational value of shellfishing. Hayes et al. (1992) used the contingent valuation method to estimate benefits to recreational shellfishing from

[18] Sedentary fishery resources such as oysters and mussels have long been subject to property rights. Sergius Orata reportedly cultivated oysters in Lake Lucrine during the early Roman empire (Bolitho 1961).

improvements to the water quality of Upper Narragansett, after expenditures for infrastructure to reduce pollution by Rhode Island communities. Survey respondents were asked their net willingness to pay for acceptable swimming and shellfishing (versions of the survey asked for swimming and shellfishing separately or combined). The contingent valuation method was also used by Damery and Allen (2004) to estimate the value of recreational shellfishing on Cape Cod, Massachusetts. Survey questions were posed to elicit the shellfisher's willingness to pay to obtain the right to shellfish, or, alternatively, their willingness to accept compensation to give up their right to shellfish (in a case where they already own a shellfish permit). English (2008) applied the travel cost method to assess the value of recreational shellfishing in southeastern Massachusetts. In addition to differences in travel costs, English's model accounted for differences in annual recreational license fees that communities impose for access to local shellfish beds. The average per-trip value for recreational shellfishing was estimated to be $21.40.

As discussed in Section 2.1, subsistence shellfishing continues to be important to many coastal Tribes and First Nations in North America. Attaching a dollar value to wild food harvests is difficult, as subsistence products do not circulate in markets. However, if families did not have subsistence foods, substitutes would have to be purchased. A cost-based method that estimate a replacement expense per pound based on the price of store-bought substitutes of a similar nutritional content can be used to derive a replacement value of wild shellfish harvests. The World Bank (2000) used this methodology to estimate the value of subsistence fisheries in Pacific Island economies.

3.2.2 Regulating Services

3.2.2.1 Water Quality Improvements

As natural biofilters that improve water quality by removing suspended solids and nutrients and lowering turbidity, shellfish are analogous to wastewater treatment facilities (Grabowski and Peterson 2007). This section discusses two methods that have been used to estimate the water quality improvement benefits of shellfish enhancement.

An estimate of the benefits generated by a regulating ecosystem service can be derived from the values of the human activities that are supported and protected by the ecosystem service (National Research Council 2004). For example, the improved water quality that results from shellfish enhancement can, in turn, enhance the quality of recreational beach use, boating and fishing. However, the values of many of these human activities may not be produced and traded in the private market economy. Consequently, methods appropriate for the valuation of non-market components of ecosystem services must be used.

An alternative method of estimating the water quality improvement benefits of shellfish enhancement is the cost-based approach, which as discussed in Section 3.1.1.1, does not provide strict measures of the economic value of ecosystem services, but rather provides rough indicators of the value by assuming that, if people incur costs to avoid damages caused by lost ecosystem services, or to replace the services of ecosystems, then those services must be worth at least what people paid to replace them. For example, the nutrient reduction costs of an individual unit of oyster reef can be quantified and compared to the cost of processing a similar amount of nutrients with a wastewater treatment facility (Grabowski and Peterson 2007).

This section describing the valuation of water quality benefits concludes with a discussion of the proposed participation of shellfish enhancement projects in water quality trading programs. Nontraditional nutrient trading entities, including shellfish aquaculture and restoration, are increasingly proposed as deserving of nutrient credits (Maroon 2011). The establishment of a market in which entities are free to buy and sell nutrient credits or offsets among themselves would generate a market price for nutrient reduction, and thereby place a monetary value on the ability of an ecosystem, such as an oyster reef, to improve water quality.

Rose et al. (2012) caution that the search for innovative nitrogen removal pathways and technologies commences with very limited knowledge of the costs and feasibility of alternatives. Given this lack of understanding, the authors state that the challenge is to create incentives and a competitive process whereby people can test the costs and effectiveness claims of alternative nutrient-removal technologies against the hard reality of experience. Even if it demonstrated that shellfish enhancement can make a significant contribution toward improving water quality, Cerco and Noel (2007) emphasize that it will not be a panacea for the host of problems associated with nutrient overload in coastal waters. For example, Peter-Contesse and Peabody (2005) point out that the input of nutrients in many parts of Puget Sound far exceeds the processing capacity of filter feeders. On the other hand, filtering by shellfish may increase the water quality benefits resulting from other management actions such as wastewater treatment, land use changes and nonpoint pollution controls (Henderson and O'Neil 2003). Furthermore, shellfish enhancement and other bioremediation measures may have cost advantages by their direct impact on eutrophic coastal zones and the multifunctional abatement of several pollutants (Gren et al. 2009). As Burkholder and Shumway (2011) state, shellfish aquaculture not only can reduce nutrient loads and phytoplankton blooms, it can also remove toxic contaminants and reducing concentrations of microbial pathogens.

Benefit Valuation Methods

As noted above, the value of the water quality improvements obtained from shellfish enhancement can be estimated by measuring the recreational and other benefits derived from those improvements. For example, shellfish enhancement projects may increase the desirability for swimming and other water contact activities by improving water quality conditions through filtering out phytoplankton and fine sediment from the water column (Henderson and O'Neil 2003).

Numerous studies have estimated the value of the recreational benefits of water quality improvements using methods appropriate for the valuation of non-market components of ecosystem services. For example, Freeman (1995) reviewed studies of valuation of water quality improvement and various types of marine recreation, including sport fishing, swimming and related beach activities, and boating. This literature employed several revealed and stated preference methods, and the review showed that there is substantial variation in value measures across studies. The author attributed these differences to five sets of factors: intrinsic differences in the resource, that is, differences in sites (or species in the case of fishing); differences in the socioeconomic characteristics of the populations whose values have been estimated; differences in the way in which sites and markets have been defined; differences in the economic models of behavior and choice on which estimates are based (including functional forms of demand functions); and differences in the econometric estimation techniques used. As Freeman notes, the last three sets of factors involve choices made by the investigator in the course of designing and carrying out the study.

While numerous studies have been conducted on determining the value of the recreational benefits of water quality improvements, only a few of these studies focus specifically on the improvements likely to result from an enhanced population of shellfish. Hicks (2004) used a travel cost model of recreation demand to analyze the economic benefits to Chesapeake Bay's recreational fishermen from proposed oyster reef restoration programs. The model explicitly links historical oyster bottom conditions to recreational fishing catch to capture ecosystem and habitat benefits. Hicks determined that recreational anglers would realize benefits of approximately $640 thousand per year for restoring 1,890 acres at 73 reef sites in the Bay. The total cost of the restoration was determined to equal $27.0

million. When calculating the net present value of the 30-year stream of benefits, assuming a discount rate of three percent, it was determined that the recreational benefits would equal approximately half of the cost of the restoration.

Cost-based Methods

Following the methodology outlined by King and Mazzotta (2000c), the first step of a cost-based approach is to quantify the nutrient removal capacity of shellfish enhancement projects. This assessment would determine the current level of water quality, and the expected level of water quality after shellfish enhancement. As discussed above, researchers are applying theoretical models to assess the nitrogen reduction potential of shellfish enhancement.

The next step is to estimate the costs of providing a substitute or replacement for the nutrient removal services of shellfish enhancement. The dollar values of providing replacement nutrient reduction services by, for example, a wastewater treatment plant can be compared to shellfish aquaculture or restoration costs to determine whether shellfish enhancement is worthwhile in terms of water quality improvements. This comparison may be presented in terms of per unit nutrient reduction costs (Lindahl et al., 2005). The replacement value is calculated as the savings in costs from using shellfish enhancement instead of more costly nutrient abatement options.

Several studies have used cost-based methods to estimate the water quality improvement benefits of shellfish enhancement, although not all of the studies follow the above methodological steps. In an early application of the cost-based method, Newell et al. (2005) used U.S. Environmental Protection Agency estimates of the cost of reducing nutrient inputs necessary for meeting the water quality goals in the Chesapeake Bay. After converting this cost figure to an average per kg cost of removing nitrogen, Newell et al. used modeled nutrient reduction estimates to derive an annual replacement value of nitrogen removal by oyster reefs in the Choptank River, Maryland of $314,836, or $181 per hectare. The authors postulated that over the ten-year life span of the oysters the value of the nitrogen removal by the oyster reefs increased to $3.1 million, or over twice the dockside value of those same oysters.

Ferreira et al. (2007) used modeling results for a hypothetical 6,000 m^2 oyster farm to show an annual net removal of around 10,683 kg of nitrogen. If a standard population-equivalent (PEQ) of 3.3 kg of nitrogen per person per year is considered, the net nitrogen removal from the farm corresponds to an untreated wastewater discharge from over 3,000 PEQ. While noting that wastewater treatment costs per inhabitant are highly variable, the authors assumed an average unit treatment cost of about €650. The estimated replacement value of the oyster farm as regards nutrient removal is about €2 million per year, almost half of the estimated combined total annual income of the farm operation. In another modeling exercise, Ferreira et al. (2011) estimated that in a 11.4 hectare Manila clam farm on the southern coast of Portugal, shellfish filtration provides a net nitrogen removal of about 28,867 kg per year, which equates to the emissions of 8,748 PEQ. The authors estimated the replacement value of the clam farm to be about €0.26 million per year, about 10 percent of the direct income of the farm.

Using production and nutrient removal data to calculate the role of all European Union shellfish farms in removing nutrients, Ferreira et al. (2009), estimated that the farms removed a total of over 55,000 tons of nitrogen per year, that is, a PEQ of 17 million people, or the population of the Netherlands. The estimated replacement value is estimated to be €0.4 billion per year. Based on the worldwide reported shellfish aquaculture landings and modeling results of nitrogen removal for a typical range of cultivated species, the authors estimated a global replacement value of €7.5 billion.

Using a close approximation of the cost-based methodology described by King and Mazzotta, Gren et al. (2009) estimated the replacement value of nutrient cleaning by mussel cultures in the Baltic Sea using a nonlinear programming model that compares costs and impacts of mussel farms with other nutrient abatement measures such as wastewater treatment plants, changes in land use and fertilizer practices, increased cleaning by households and industries not connected to municipal wastewater treatment and creation of wetlands. The results indicate that calculated marginal cleaning costs of nutrients by mussel farming can be considerably lower than other abatement measures, with cost savings between 2 and 11 percent. The authors note that the cost-effectiveness of mussel culture in nutrient removal from the water column depended on mussel growth, sales options, assumptions about mussel farming capacity and the nutrient load targets.

Stadmark and Conley (2011) reported that compared to other potential nutrient abatement measures, mussel farming is a relatively expensive measure. Their calculations show that nitrogen reduction in mussel farms (industry mussels) is about two times more expensive per kg nitrogen than nitrogen reduction from wastewater treatment plants. However, Rose et al. (2012) point out that in estimating the cost of nitrogen removal, Stadmark and Conley did not count the revenue from the sale of shellfish biomass as an offset to costs. Rose et al. note that other studies (e.g., Gren et al. 2009; Miller 2009) have shown that the cost of nitrogen removal from shellfish production is highly contingent upon the returns that could be achieved through shellfish biomass (either for animal or human consumption). In addition, Rose et al. note that the costs quoted by Stadmark and Conley fail to reflect the wide range of costs (and cost uncertainty) associated with nitrogen removal, and that they substantially understate the upper bound limits of nitrogen removal costs from wastewater treatment plants. According to Rose et al. note, extensive evidence elsewhere demonstrates that these costs can increase rapidly as treatment approaches limits of technology and as smaller plants add tertiary nitrogen removal processes.

Water Quality Trading

Water quality trading is a "market-based" approach to achieving water quality standards within a defined area.[19] It allows a source discharging nutrients to the coastal zone to meet its regulatory obligations by using nutrient reductions created by another source that has lower pollution control costs (U.S. Environmental Protection Agency 2003). The economic argument for this approach is that it is able to achieve the resulting environmental outcome more efficiently (i.e., at a lower unit cost of abatement) than traditional "command-and-control" approaches because it gives sources the flexibility to search for the lowest cost pollution reduction option.[20] These nitrogen reductions are also called offsets because the reductions are made in order to compensate for or to offset an emission made elsewhere.

One potential option available to sources is the purchase of nutrient (e.g., nitrogen or phosphorus) credits from shellfish enhancement projects which are nutrient sinks (Ferreira et al. 2007). For example, the costs for controlling nitrogen loads at a point source such as a wastewater treatment plant compared to restoring shellfish production can vary significantly, creating the impetus for water quality trading. Through water quality trading, municipal and industrial wastewater treatment plants that face higher nitrogen removal costs—particularly as nitrogen concentration limits are tightened—could meet their regulatory obligations by purchasing "pollutant reduction credits" from,

[19] There are two basic forms of emission trading systems: cap-and-trade systems that based on an initial allocation of emission rights through a regulatory body, and baseline-and-credit systems (also known as offset systems) where emission credits are generated through the implementation of an activity which reduces emissions against a baseline (Profeta and Daniels 2005). The water quality trading approach discussed in this report is a baseline-and-credit system. The U.S. Environmental Protection Agency's Acid Rain Trading Program controlling sulfur dioxide emissions is an example of a cap-and-trade system.

[20] Under a command and control approach, the regulated entities are given little discretion in their pollution control efforts. For example, the government may mandate what pollution-reduction technologies they should use.

say, a shellfish aquaculture operation if the cost is lower and the nitrogen reductions achieve targets. The establishment of a market in which nutrient credits or offsets are freely traded can generate a market price for nutrient reduction. This market price represents the interests of wastewater treatment plants affected by nitrogen concentration limits to reduce nitrogen emissions in the most cost-effective manner.

Nitrogen credit trading at the watershed scale is now a reality in parts of the United States (Ferreira et al. 2011). In 2003, the U.S. Environmental Protection Agency issued a national water quality trading policy to support the development and implementation of trading in water quality management (U.S. Environmental Protection Agency 2003). The agency advocates water quality trading as a cost-effective means to preserve and improve water quality, and a number of programs are in various stages of development. While the policy doesn't specifically refer to shellfish enhancement as part of a nutrient trading scheme, it does support the creation of water quality trading credits in ways that achieve ancillary environmental benefits beyond the required reductions in specific pollutant loads, such as the creation and restoration of wetlands, floodplains and wildlife and/or waterfowl habitat. Using shellfish enhancement as part of a nutrient trading scheme would fulfill this objective, as shellfish enhancement would provide multiple economic and environmental benefits (Golen 2007).

In 2002, the Connecticut Department of Energy and Environmental Protection developed the Nitrogen Credit Exchange among 79 wastewater treatment plants to improve water quality in Long Island Sound, with over $30 million in economic activity in the first four years of trading (Ferreira et al. 2011). Similarly, the Chesapeake Bay states are developing water quality trading programs to provide regulated sources options to remain in compliance with their individual nutrient wasteload allocations, permit requirements or aggregate mass load caps (Maroon 2011; Stephenson and Shabman 2011).

Furthermore, a number of U.S states and countries outside of the United States are considering the use of shellfish in nutrient trading schemes in addition to other nitrogen reduction measures. As such, researchers are applying theoretical models to assess the nitrogen reduction potential of shellfish farms and for the valuation of nitrogen credits (Getchis and Rose 2011). The nitrogen reduction costs directly associated with a shellfish enhancement project are the project's costs of production, including the costs of labor and capital (see Section 4.1). If the shellfish produced by a project can be sold on the open market, the income from these sales can offset project costs (Stadmark and Conley 2011). The costs for nitrogen reduction measures also depend on the amount of nitrogen removed during the harvesting of shellfish (Stadmark and Conley 2011). As mentioned in Section 2.5, the nitrogen reduction potential of shellfish enhancement projects is the subject of intense (sometimes controversial) research.

Brumbaugh and Toropova (2008) and Lindahl et al. (2005) postulate that if a well-designed and regulated market existed for trading the nitrogen removed by shellfish, it should have the effect of spurring further investments in shellfish enhancement. By providing shellfish enhancement projects an opportunity to earn additional revenue when they sell nutrient reduction credits, a trading program allows projects involved in culturing shellfish and restoring beds to capitalize on the nitrogen removal services provided by their activities. However, Stephenson and Shabman (2011) note that participation of shellfish enhancement projects in a trading program is contingent upon buyers (e.g., wastewater treatment plant) and sellers (e.g., shellfish growers) reaching an agreement on the amount of credits to be provided to the buyer, how it will be documented that the credits were provided and the payment terms for the provision. Stephenson and Shabman suggest that these agreements will be described in a contract between the buyer and seller. The contract also will assure the buyer that the conditions set by the regulatory authorities are being met by the seller so that the credits can be used as offsets in the trading program. The authors note that nitrogen and other nutrient offsets lend themselves to measurement and verification, making credits especially amenable to a contracting process where payment is contingent on demonstrated results (Stephenson and Shabman 2011). In a similar vein, DePiper and Lipton (2011) discuss how an enforced contract can result in a switch in how an oyster reef is managed, e.g., in the absence of a contract, oysters may be harvested more aggressively, and consequently the desired nutrient removal levels may not be achieved.

In the aforecited article, Stephenson and Shabman also suggest that a time referenced benchmark would appear necessary in order to prevent an existing shellfish enhancement project to claim credits for past investments. Once such time referenced baselines are established, expansions of nutrient credit services (e.g., expanded oyster aquaculture production beyond the referenced date) could be counted new (additional) services and credited.

Some researchers are somewhat circumspect about participation of shellfish enhancement projects in nutrient trading schemes. Golen (2007) raises the issue of how to satisfactorily define and measure nitrogen removal when using shellfish. As discussed in Section 2.2.1, shellfish enhancement can remove nitrogen from the water column in two ways: 1) removal of nitrogen incorporated in the shell and tissues during harvesting and 2) the action of denitrifying bacteria on shellfish biodeposits (pseudofeces and feces). Golen notes that while it is relatively easy to account for nitrogen removed in shell and flesh based upon annual harvest levels, the factors that govern the magnitude of nitrogen removal via denitrification are too complex and variable to allow the use of fixed removal rates that can be applied across all shellfish aquaculture facilities. Nitrogen trading using shellfish could proceed based solely upon the quantification of nitrogen removal via biomass at harvesting, but this could substantially underestimate the nutrient reduction value of shellfish enhancement: nitrogen removal by oysters via nutrient burial and denitrification of biodeposits is estimated to be seven times the weight of nitrogen contained in the shellfish meat (Stephenson and Brown (2006); cited in Golen (2007)).

Stadmark and Conley (2011) suggest that the primary focus for nutrient mitigation should be on nutrient reductions from land-based sources before nutrients enter into the coastal zone. However, Rose et al. (2012) argue that this view may be somewhat simplistic. Rose et al. agree that nutrient mitigation should not be used in lieu of land-based nutrient reductions, but they note that as conventional treatment methods approach limits of technology (wastewater treatment plant upgrades) or become very expensive (urban stormwater retrofits), many estuaries will still need to find additional nutrient removal methods to meet water quality objectives.

Lindahl and Kollberg (2009) support shellfish enhancement as a nutrient compensation measure but argue that it should be reserved for diffuse (nonpoint source) emissions into the coastal zone coming from, for example, agriculture, because for these there are few other effective options available. Rose et al. (2012) also note that nutrient removal by shellfish becomes more important as the main nutrient loading from land increasingly shifts to diffuse sources. According to the authors, as the majority of point source contributions are reduced, the law of diminishing returns makes it far more expensive to reduce the remainder of the nitrogen load to coastal ecosystems, largely attributable to agricultural and diffuse urban sources.

While shellfish enhancement projects have yet to become active participants in a U.S. nutrient trading program, a number of studies have examined the credit generation potential of these projects. Stephenson et al. (2010) describes a bioeconomic model used to estimate the cost of providing a nitrogen offset from commercial oyster aquaculture. The model estimated cost by calculating the supplemental compensation necessary for a commercial aquaculture operation to produce additional oysters. The study calculated total nitrogen removed from ambient waters based on the amount of nitrogen sequestered in oyster tissue and shell at harvest and nitrogen removed through denitrification of oyster biodeposits. The estimated annual cost to generate one pound of nitrogen offset ranged from $0 to $150, depending on assumptions about the market price of harvested oysters, project input costs, and oyster growth and mortality rates. According to the authors, conceptually, costs may be zero because in some economic circumstances oyster growers would be willing to expand production without any compensation for nutrient removal services. Estimated total nitrogen removed for every one million market-sized oysters harvested ranged between 290 and 815 lbs per year, depending on assumptions of denitrification rates.

Piehler and Smyth (2011) compared the ecosystem service value of nitrogen removal in five of the major habitat types found in temperate estuaries (salt marsh, submerged aquatic vegetation, oyster reef, intertidal flat and subtidal flat). The authors estimated a dollar value of nitrogen removal using the rates from the North Carolina nutrient offset program. Under this program, the nutrient offset payment price is not set by the market but rather is a regionally derived number that had significant stakeholder input in its determination. The trading price of the North Carolina nutrient offset program at the time of the study was $13 per kg of nitrogen removed. To best estimate the annual value of habitat specific nitrogen removal, the authors multiplied this price by the mean annual rates of denitrification and the standard error of these means. The authors estimated that submerged aquatic vegetation and oyster reefs provide about $3,000 per acre per year in nitrogen removal services, while wetlands provide about $2,500 per acre per year of nitrogen removal service.

Outside of the United States one can find examples of shellfish enhancement projects being incorporated into nutrient trading schemes. For instance, Lindahl and Kollberg (2009) described what they called the first case in Sweden of trading a nitrogen emission. The Swedish community of Lysekil was permitted, as a trial between 2005 and 2011, to continue to emit nitrogen from its wastewater treatment plant into an estuary provided that the same amount of nitrogen was ''harvested'' and brought ashore by 3,900 mt of farmed blue mussels. According to the authors, the payment of €150,000 that Lysekil made to the mussel farming enterprise contracted for the nitrogen removal was far below the cost for nitrogen removal in the local wastewater treatment plant.

3.2.2.2 Shoreline Protection and Stabilization of Submerged Land

As described in Section 2.2.2, the physical structure of an oyster reef can protects shorelines and inland waters from the erosive force of waves. Henderson and O'Neil (2003) suggest that in the absence of other data, cost-based methods could be used to estimate the economic benefit of an oyster reef for shoreline protection or sediment stabilization. For example, cost-based methods might answer the questions, "if the oyster reefs weren't there, are there structures or other ways to provide the same services the reefs perform?" and "what is the cost of providing the substitute?"

Previous efforts to protect shorelines have largely involved constructing bulkheads and seawalls (National Research Council 2007a). Both of these methods have been shown to cause vertical erosion down the barrier, subsequent loss of intertidal zone and even increased erosion on adjacent properties (Scyphers et al. 2008). The benefits of oyster reefs as an alternative to the hard bulk-heading of shorelines have not been valued in the literature. However, Conservation International (2008) discusses the results of several studies that analyzed the economic contribution of shoreline protection services provided by coral reefs based on estimates of the extent of shoreline protected by coral reefs and the costs required to replace the reefs by artificial means.

As discussed in Section 3.1.1.1, cost-based methods do not provide strict measures of the economic value of ecosystem services, and may misrepresent the value in certain circumstances.

3.2.2.3 Property Values

Changes in the value of waterfront property may reflect the full range of positive (and negative) impacts of shellfish enhancement, including local changes in environmental amenities such as water quality and shoreline protection. Consequently, any expected effects on those other factors may be expressed in terms of effects on property values. Estimating the value accruing to private tideland owners is an especially critical part of the balance sheet for shellfish enhancement in states such as Washington, where 75 percent of tidelands is owned privately (Peter-Contesse and Peabody 2005). Moreover, the added value from waterfront properties not only has implications for added wealth for the buyers and sellers of these properties; it also has value to local communities and states through accompanying real estate taxes and property transfer taxes in many states (Kildow 2008). Yet, in comparison to activities directly benefiting from restoration projects such as recreational fishing and beach use, real estate values have received far less attention from researchers (Kildow 2008).

Kildow (2008) discussed a number studies that use various economic valuation methods to examine how coastal preservation and restoration influence housing prices for waterfront homes. Included in this literature survey are two hedonic property studies that demonstrate the impact of water quality on housing values along the Chesapeake Bay; one study was conducted in Anne Arundel County, Maryland (Leggett and Bockstael 2000), and the other was conducted in the St Mary's River Watershed (Poor et al. 2006). A third study by Braden (2004) that combined two methods, the hedonic method and conjoint analysis, used housing market data and also measured the preferences of homeowners in Lake County, Illinois.

The effect of shellfish enhancement on the value of waterfront property may not necessarily be positive. As aesthetic values associated with shorefront property have increasingly become more important, the so-called not-in-my-backyard (NIMBY) factor has come into play: waterfront property owners do not want their views affected by commercial ventures, including shellfish aquaculture operations (National Research Council 2010). This attitude is most likely to occur in communities or settings where shellfish aquaculture has not been part of the established or traditional waterfront or where there is a large influx of residents who do not share the community's cultural fishing traditions (National Research Council 2010). Landowner opposition may also arise when changes occur in the species cultivated that result in the use of more intensive culture methods (Dewey et al. 2011). Shellfish growers can design their operations to minimize visual and water access conflicts, and it is possible that prospects for permitting of shellfish enhancement projects can sometimes be improved by educating the local community about the ecological benefits (e.g., water filtration, nutrient removal, habitat enhancement for valuable finfish and crustaceans) of those projects (National Research Council 2010). However, anecdotal information presented in a number of studies (e.g., Kite-Powell 2009; Northern Economics 2010) suggests that shellfish aquaculture operations may lower the value of real estate in some coastal communities. As will be discussed in Section 4.2, any such decrease in property value should be considered a shellfish enhancement cost.

3.2.3 Supporting Services

As discussed in Section 2.3, the structure provided by shellfish reefs and beds serves as habitat for other species of fish and crab, and protects the ecological integrity of other habitats, such marshlands and seagrass beds, which also support a wide variety of marine life. These improvements may ultimately lead to measurable increases in the production of additional types of finfish and invertebrates targeted in commercial and recreational fisheries. Larger populations of important commercial and recreational species potentially mean significant contributions to economic welfare in the form of greater industry profits and consumer benefits.

Few studies have yet sought to quantify secondary production attributable to shellfish enhancement. Peterson et al. (2003) reviewed available empirical data on quantitative enhancement of nekton populations by restoring oyster reefs in the southeast United States and applied demographic and growth models to estimate the species specific augmentation of fish and crustacean production that is expected per unit area of oyster reef restoration. They estimated that 10 m^2 of restored oyster reef yielded an additional 2.6 kg per year (2,600 kg per hectare per year) of production of fish and large mobile crustaceans for the functional lifetime of the reef. Grabowski and Peterson (2007) converted the amount of augmented production per each of the species groups that were augmented by oyster reef habitat in Peterson et al. to a commercial fish landing value. According to the researchers, for fish of commercial significance, the enhanced production by the reefs equates to $3,700 per hectare per year and, over a 50 year time span, the fish productivity would exceed the anticipated value of directed oyster harvest from the same area by more than 34 percent.

A study in North Carolina (West Bay, Neuse River) by Lenihan and Grabowski (1998) compared the value of fish and crab from three oyster reefs to the value of harvest from adjacent unstructured sand bottom areas. A total of 15 commercially valuable species were found to utilize restored oyster reef habitat. The study results indicated that the long-term commercial value of these fish and crabs was greater than the value of the oyster production.

Recreational anglers who are aware of the species variety and abundance of fish available over oyster reefs value the reefs for the enhanced recreational fishing opportunities they provide. Isaacs et al. (2004) employed the contingent valuation method to estimate the value of Louisiana's oyster reefs as recreational fishing grounds, using a sample drawn from resident saltwater anglers who participated in the National Marine Fisheries Service's Marine Recreational Fishing Statistical Survey. A telephone survey was conducted featuring a dichotomous-choice net willingness to pay question. The average annual net willingness to pay among resident saltwater recreational fishermen to maintain access to recreational fishing over Louisiana's oyster reefs was $13.21.

3.2.4 Cultural Services

Wilson and Farber (2008) note that opportunities for recreation and natural amenities (e.g., productive sport fisheries, clean beaches) get an inordinate amount of attention in the economic literature, while other benefits such as aesthetic, spiritual and historic do not get much attention at all. Although a number of qualitative descriptions of the culture services provided by restored, enhanced or managed shellfish populations appear in the literature (see Section 2.4 for examples), no example of an attempt to measure the benefits of these services in monetary terms could be found.

3.2.5 Non-Use Values

As described in Section 3.1.1.2, non-use values include existence value and option value. For example, Hicks et al. (2004) suggested that people may benefit from oyster reefs in Chesapeake Bay even if they do not directly use the environmental asset by either deriving value from knowing that oyster reefs exist and provide ecosystem services (existence value) or from knowing that improved environmental conditions might make future use of the Bay more enjoyable should they choose to use the Bay directly (option value). To measure these non-use values Hicks et al. used the contingent value method. Based on the results of a mail survey of residents of New Jersey, Delaware, Maryland, Virginia, and North Carolina, the researchers determined that the non-use value of a ten-year oyster reef project, consisting of 10,000 acres of oyster sanctuary and 1,000 acres of artificial reef, to be at least $14.91 per household per year with a median estimate of $86.86 per household per year. Aggregating to the general population, the researchers estimated the total non-use value of the project to be at least $114.95 million.

3.3 Avoiding the Double Counting of Benefits

The above sections have examined appropriate methodologies for estimating the benefits arising from various categories of ecosystem services provided by shellfish enhancement. However, the benefits estimated by each methodology can not necessarily be added to arrive at a measure of the overall value of such projects—there may be some overlap between values estimated by different methodologies.

For example, the recreational value of improvements in the quality of coastal waters may be measured using travel cost methodologies, yet may also influence the value of local homes (e.g., residents may wish to live in close proximity to valued recreational resources), thereby influencing hedonic property value estimates. Moreover, a portion of the recreational value may be reflected in residents' net willingness to pay for water quality improvements (Johnston et al. 2002). In these and many other cases, summing of value estimates from different methodologies may double-count the same values. Analysts must consider this potential for overlap when developing a comprehensive, integrated approach to assessing the benefits of shellfish enhancement.

4 Costs of Shellfish Enhancement

Not all the effects of shellfish enhancement are necessarily beneficial. An economic analysis must consider the costs as well as the benefits, where cost is represented by the value of the next best alternative foregone as the result of making a decision, i.e., the opportunity cost. This section describes the potential costs that may be incurred in efforts to protect or restore a shellfish population. A summary of these potential costs is provided in the table below.

Table 3 Costs of Shellfish Enhancement

Project costs	Start-up and operating costs (capital, labor, fuel, seed, etc.)
Environmental costs	Aquaculture (changes in community composition, habitat transformation, genetic impacts on wild populations, coastal use conflicts, etc.) Introduced shellfish species (trophic interactions, habitat transformation, interactions with native species, etc.)
Human health costs	Shellfish poisoning (lost wages and work days, medical treatment, poisoning cause investigations, shellfish monitoring programs, etc.)

4.1 Project Costs

As discussed in Section 3.2.1.1, several studies have examined the costs (as well as the revenues) of commercial shellfish fisheries and aquaculture operations. However, Henderson and O'Neil ((2003) note that the costs incurred in designing and implementing shellfish restoration projects have been more difficult to fully account for because they are typically borne by public entities. The authors also observe that the opportunity costs (e.g., spending public monies for other purposes) and secondary and higher order costs are impossible to completely track. Restoration project costs include several major elements: planning and project development; the cost of eyed oyster larvae and remote setting equipment; bottom cultch and placement material; other equipment and maintenance; transportation and vehicle costs; a baseline survey; the cost of a monitoring program; data analysis and interpretation; enforcement; infrastructure; labor; and project implementation (these costs reflect the management personnel expense needed to manage project activities) (Industrial Economics 1999).

As noted in Section 1.2, Brumbaugh et al. (2007) indicate that even relatively small restoration projects can be costly (e.g., > US $100,000 per acre for restored oyster reef). According to The Nature Conservancy (2008), the surveying and up-front permitting requirements alone can be extensive, lengthy and costly. Project costs vary substantially with the purpose of the project as well as the size. For example, Henderson and O'Neil (2003) note that Maryland and Virginia both constructed oyster reefs in Chesapeake Bay using base material of oyster shell. The cost of shell base materials for both the Virginia and Maryland reefs was about $10,000 per acre. However, the Maryland reefs were also initially seeded at a cost of $10,000 per acre and maintained annually by addition of broodstock and more shell base material, creating a "put and take" fishery. The initial stocking and addition of maintenance broodstock resulted in higher productivity, but at an increased cost.

4.2 Environmental Costs

Though to a far lesser extent than finfish, shellfish aquaculture can also have undesirable effects on the environment (Sequeira et al. 2008; National Research Council 2010). It may result in changes in benthic community composition through a range of mechanisms, such as excessive partitioning of food resources, competition for space and increased sediment deposition (Sequeira et al. 2008). For example, Bendell-Young (2006) found that intertidal regions in British Columbia that had been used for intensive farming of Manila clams, *Venerupis philippinarum*, were characterized by a decrease in species richness, altered species abundance and distribution, change in community structure composed of surface species, sub-surface species and bivalves, to one composed primarily of bivalves, and greater accumulations of surface sediment silt and organic matter.

Submerged and floating shellfish cultivation gear may also have negative impacts on essential marine habitats. For example, the physical disturbance caused by oyster cultivation gear may cause deterioration of eelgrass beds, an essential habitat for juvenile fish and shellfish (Getchis 2005; National Research Council 2010), and physical alteration to prospective geoduck aquaculture sites through grading and rock removal may result in damage to ecological functions (Washington Department of Ecology 2009). Mechanical harvesters, commonly used for scallops and clams, can create significant environmental stress, damaging the ocean floor and harming benthic plant life and other wild species (Brumbaugh et al. 2006; Stokesbury et al. 2011; SeaChoice undated). However, harvesting of culture plots with mechanical harvesters is less destructive than harvesting wild shellfish because harvest is restricted to relatively small plots (SeaChoice undated).

In addition, there is the risk of contaminating wild shellfish populations with cultured genes. With culture of a native species, this risk centers on the potential loss of natural genetic variation, which serves to buffer the population against natural selective forces (Straus et al. 2008). In some aquaculture operations, "seed" (juveniles) can be taken from the wild; however, seed scarcity has forced other operations to turn to hatcheries (Cross et al. 2008). For example, geoduck growers obtain seed from hatcheries. Since the geoducks planted by aquaculture operations may reproduce before harvest and all bivalve molluscs are broadcast spawners, there is a potential for the cultured clams to interact with the genetics of the wild populations (Washington Department of Ecology 2009). Moreover, diseases and parasites carried by hatchery seed may be introduced into areas where they currently do not exist, with a consequent deleterious effect on wild shellfish populations (Cross et al. 2008; Straus et al. 2008; Washington Department of Ecology 2009).

The extensive use of nets, docks, cages and other gear by shellfish farms may also conflict with navigation, dredging, commercial and recreational fishing, swimming and other users of coastal waters and the adjoining shoreline. Moreover, shellfish farms may have to compete for space with residents who have shoreline vistas. Farms are usually hidden below the waterline, but at low tide much of the gear is exposed (Thacker 2006).[21] In a review of aquaculture siting issues in Washington State's coastal zone during the 1970's, Evans (1979) reports that among the complaints expressed by concerned citizens before the Kitsap County Board of Commissioners was that hydraulic harvesting of subtidal hardshell clams destroys subtidal aquatic environments and associated plants and animals; causes extraordinary amounts of broken clam shells to be washed up on nearby beaches; results in unacceptably high levels of turbidity (amount of suspended solid materials in the water); and is unacceptably noisy and interferes with the residential character of the area. At the time Evans' report was written, harvesting of hardshell clams was only one example of a growing controversy over

[21] Sommers and Canzoneri (1996) note that Puget Sound growers have been increasingly using floating aquaculture structures. According to the authors, floating aquaculture is often associated with heightened use conflicts because it occupies surface water and allows more intensive growing operations.

aquacultural use of aquatic areas in Washington. Conflict also developed over the siting and development of mussel rafting operations in Island and San Juan Counties and oyster farming in Pierce County and Hood Canal (Evans 1979).

More recently, the conversion of some of Washington's intertidal beaches to geoduck aquaculture has resulted in conflicts with a number of shoreline residents who feel geoduck aquaculture alters the nature of their shorelines (Coalition to Preserve Puget Sound Habitat 2007). Some shoreline residents dislike having what they see as an industrial activity occurring near them, and many people are concerned that geoduck aquaculture will harm the ecological functions of the shorelines (Washington Department of Ecology 2009).

Because shellfish are ecosystem engineers, translocations of native species and introductions of nonnative species can have disproportionately high environmental impacts, many of which are potentially undesirable (Ruesink et al. 2005). Such translocations and introductions have been employed to augment or replace depleted natural stocks or to diversify the number of species used in aquaculture operations (National Research Council 2010). Non-native shellfish species that are brought into new environments may indeed provide such valuable services as water column filtration, habitat creation for non-shellfish species and stabilization of estuarine sediments, but they can also may compete with native species, negatively affect food webs and bring in new diseases and other undesirable species (Brumbaugh et al. 2006).

Competition between native and introduced shellfish is expected to be most intense if they share similar habitat. For example, the suminoe oyster was identified as a candidate for introduction to Chesapeake Bay because its salinity and temperature requirements closely match those of the eastern oyster; therefore, the two species would be likely to occupy the same habitat (U.S. Army Corps of Engineers 2008). Researchers have concluded that the risk is moderate to high that suminoe oysters would interact and compete with eastern oysters (U.S. Army Corps of Engineers 2008). Ruesink et al. (2005) indicate that on the western coast of North America, the native Olympia oysters tend to occur at lower depths with less temperature stress than does the introduced Pacific oysters. The authors note that it is reasonable to predict that Pacific oysters would occupy a higher tidal elevation than do Olympia oysters, and that, in places where Pacific oysters reached high density, they would transform habitat and increase epifaunal diversity. Thus, Pacific oysters would perform a novel ecosystem role in western North American estuaries (Ruesink et al. 2005). Moreover, Olympia oysters filter food particles that are smaller than those taken by Pacific oysters and, thus, serve slightly different ecological roles in controlling phytoplankton blooms (Peter-Contesse and Peabody 2005).

However, Ruesink et al. (2005) observe that competition between oyster species also occurs indirectly through habitat modification. Recent evidence suggests that Olympia oyster larvae disproportionately settle in areas with large accumulations of shell. Because intertidal Pacific oysters comprise most of the shell habitat in the bay, the native oysters only have the option of recruiting to zones where immersion times are too short for survival. Thus, the introduced oyster has developed into a recruitment sink for natives, particularly in the absence of remnant subtidal native-oyster reefs (Ruesink et al. 2005).

Species brought in with shellfish aquaculture can present problems for the continued production of the cultured shellfish species in addition to potentially interacting with native species and altering the structure and function of surrounding communities and ecosystems. For example, Ruesink et al. (2005) report that *Batillaria attramentaria*, an Asian snail introduced to the U.S. West Coast with the Pacific oyster, outcompetes the mud snail *Cerithidea californica*, which has caused local extinction of the native snail in a number of estuaries. In addition, the shell-boring sabellid polychaete, *Terebrasabella heterouncinata*, also introduced with the Pacific oyster in California, infested cultured red abalone, *Haliotis rufescens*, with great economic consequences to growers before it was successfully eradicated. Based on their review of the unintended consequences of shellfish introductions, Ruesink et al. (2005) conclude that the potential ecological costs of the deliberate introduction of shellfish suggest that native shellfish are a better option for ecosystem restoration.

4.3 Human Health Costs

Because they are capable of filtering large volumes of water relative to their size, shellfish may ingest and concentrate undesirable pathogens and other toxic pollutants in their tissues when filtering contaminated water. Although these pathogens have little effect upon the shellfish themselves, they can be harmful to the human consumer, causing diseases such as hepatitis (Busse 1998). Toxins produced by harmful algae can also be concentrated by shellfish through filter feeding to the point that the shellfish become dangerous or even deadly for humans to eat (NOAA Fisheries Northwest Fisheries Science Center and Washington Sea Grant Program 2002). Paralytic shellfish poisoning, neurotoxic shellfish poisoning, diarrhetic shellfish poisoning and amnesic shellfish poisoning are caused by eating scallops, mussels, clams, oysters and cockles contaminated with various toxins (Bauer 2006).

Shellfish enhancement projects must take into account the potential costs of shellfish consumption-related illnesses. Human sickness and death from eating tainted shellfish result in lost wages and work days. Costs of medical treatment and investigations for the cause of the sickness can also be significant. Further, individuals who are sick may experience pain and suffering (Hoagland et al. 2002). Hoagland et al. (2002) estimated the costs of various types of shellfish poisoning to be $1,400 per reported illness, $1,100 per unreported illness and $1 million per mortality. Based on these cost estimates and the estimated annual number of shellfish consumption-related illnesses and deaths in the United States, the researchers calculated that the public health costs of shellfish poisoning averaged $400 thousand per year.

In addition, the costs of shellfish monitoring programs must be considered. Most of the states in which shellfish poisoning is a significant problem operate such programs. Boesch et al. (1997) estimated that each program costs $100,00 to 200,000 per year, while Hoagland et al. (2002) estimated that the total annual cost of U.S. shellfish surveillance programs to be approximately $2 million.

Consumer concerns about the safety of shellfish consumption in regard to human health can lead to a downward shift in the demand for shellfish products and loss in consumer surplus. This loss reflects the fact that when consumers become concerned about seafood safety, the maximum amount they are willing to pay for any quantity of shellfish is less than when they are not concerned about safety (Lipton and Kasperski 2008). Keithly and Diop (2001) examined the extent to which the demand for Gulf-area oysters has been reduced as a result of mandatory warning labels and negative publicity. In general, their results suggested that since 1991 the “summer” dockside price had been reduced by about 50 percent as a result of warning labels and associated negative publicity, while the “winter” dockside price has been reduced by about 30 percent.

On the other hand, increases in demand for shellfish and gains in consumer surplus can be achieved if consumers perceive that certain shellfish products are considered safer and perhaps more environmentally-friendly (Shumway et al. 2003; Trauner 2004; Sequeira et al. 2008). A measure of the value of food safety is an individual’s net willingness to pay to reduce the health risks from eating shellfish. Lin and Milon (1995) and Lin et al. (1995) used the contingent valuation method to estimate consumers’ net willingness to pay for shellfish products that are less likely to cause illness. In both studies, a substantial number of consumers showed a positive net willingness to pay for safer oysters, and Lin et al. found that individuals who had been sick from eating unsafe oysters generally valued safer oysters more than others. The contingent valuation method was also used by Appéré and

Bonnieu (2003), who estimated how much recreational shellfish gathers would be willing to pay for a gathering site where the risk of contracting an illness is low. Instead of being asked directly to place a monetary value on the provision of a hypothetical site, respondents were asked how far they would drive to use such a site. This study also found a substantial number of consumers who held a positive net willingness to pay for safer shellfish.

Of course, if a price premium is charged in the retail market for shellfish that pose a lower health risk to consumers, the economic benefits accrue to the sellers rather than the buyers. For example, Sommers and Canzoneri (1996) state that shellfish from Washington command a premium price in the domestic market because they have gained a reputation as a superior product that is safer to eat than shellfish produced elsewhere in the United States. The authors note that this reputation rests largely on the strength of Washington's shellfish monitoring program and the emphasis that regulatory officials and shellfish growers in the state place on water quality.

4.4 Best Management Practices and Standards

The implementation of best management practices (BMPs) and regulatory standards represent one approach to reduce or minimize adverse environmental and social effects, food safety issues and other public concerns resulting from proposed shellfish enhancement projects. BMPs and standards are general overarching principles and specific procedures or methodologies used to guide the day-to-day operation of aquaculture businesses (Getchis and Rose 2011). They are often developed by the industry group (e.g., shellfish growers) to which they apply but also can be driven by regulatory agencies or other nongovernmental organizations. Adoption of and adherence to BMPs is usually voluntary, while regulatory standards usually are imposed by a public authority (i.e., federal, state, or local agencies responsible for permitting and oversight of shellfish aquaculture); compliance is often required by law as a condition of the permit (National Research Council 2010). Lists of BMPs and standards that can be applied across shellfish species and conditions are provided by Dewey et al. (2011), Hargreaves (2011) and the National Research Council (2010).

Expenditures of time or money in conforming with BMPs and standards increase the project costs of shellfish enhancement (National Research Council 2010). Some shellfish growers question whether BMPs and standards are attainable and affordable for the average aquaculture operation (Getchis and Rose 2011). However, some BMPs and standards have been proven to increase efficiency and hence profitability, while reducing environmental impacts (e.g., the use of triploid oysters in order to mitigate against successful reproduction and spread in areas where the oyster is nonnative) (National Research Council 2010). Moreover, some BMPs and certification standards for shellfish products (e.g., organic, sustainable, fair trade, domestically or even locally grown) can be used as a marketing tool to enhance the value of those products (National Research Council 2010). Hargreaves (2011) notes that product certification for shellfish aquaculture is a work in progress and provides a review of its current status.[22] Although consumer demand for "ecolabeled" seafood is growing, Hargreaves cautions that considerable uncertainty remains about the price premium, market access or other value that can be obtained by producers of ecolabeled shellfish.

The National Research Council (2010) noted that in settings where shellfish aquaculture is carried out by a number of small, independent operators, it makes sense to develop BMPs or standards that set parameters based on system-wide ecological carrying capacity (defined as the aquaculture biomass

[22] According to its website, the Aquaculture Stewardship Council (ACS), an independent, nonprofit organization, will be the world's leading certification and labeling program for responsibly farmed seafood (Aquaculture Stewardship Council 2012). The ACS certification process will allow for shellfish producers to use an ecolabel identifying the product as environmentally sustainable (Getchis and Rose 2011).

production above which unacceptable ecological impacts arise (see Inglis et al. (2000) and McKindsey et al. (2006)), thereby taking into account the cumulative effects of all farming operations. Ecological carrying capacity is typically calculated using ecosystem modeling techniques (see Section 2.5). Stakeholders (e.g., habitat and aquaculture farm managers, commercial and recreational fishermen, coastal land owners and nongovernmental organizations) can help identify ecosystem components that need to be evaluated in unbiased ecological carrying capacity assessments as well as contribute to decisions about what are acceptable or unacceptable environmental impacts (National Research Council 2010; Byron et al. 2011). According to the National Research Council (2010), BMPs or standards that target parameters related to ecological carrying capacity can be focused on shellfish aquaculture broadly and may not have to address each location, species and culture technique separately. However, given that the ability to quantify and measure ecological carrying capacity remains limited, adopting this approach will require careful consideration of the risks, the acceptable level of ecological change and the appropriate parameters to monitor.

5 Economic Valuation Issues and Considerations

5.1 Distribution of Benefits and Costs across Society

In assessing the economic value of a shellfish enhancement project, the economic efficiency criterion requires only that there be a net positive change in welfare for society as a whole in order for the project to be justified. While delivering net benefits to society as a whole is important and should be given due weight, planners and decision makers are well aware that it is only one consideration—the way in which the costs and benefits of the project are distributed within society (i.e., who receives the benefits, who pays the costs) can also be important. Typically, planners and decision makers have to consider the extent to which such distributional impacts are important and need addressing.

The benefits estimated by different economic valuation methods are, in many cases, realized by different segments of the population (Johnston et al. 2002). For example, the hedonic pricing method assesses benefits (and costs) realized by local coastal property owners, while the travel cost analysis estimates values received by both local residents and tourists.[23] The market price method, in contrast, measures economic values realized off-site by a range of resident and non-resident user groups who supply and consume shellfish products. Other ecosystem services may also occur off-site; for example, when juvenile finfish sheltered by an oyster reef are caught at a later life stage miles away, or improvements in water quality arising from an oyster reef extend far beyond the reef itself (Johnston et al. 2002; Henderson and O'Neil 2003). Still other benefits may accrue to people who receive enjoyment from simply knowing that the cultural values associated with the lifestyle of the watermen are being preserved (Wasserman and Womersley undated). In short, how certain individuals will be affected by a shellfish enhancement project that alters the flow of ecosystem services will likely depend on the type of service. Services that are actually traded in markets; services that are not marketed but consumed on-site; services that are not marketed and produce ecological effects off-site—each may have a different affected population (Plummer 2009).

5.2 Distribution of Benefits and Costs through Time

As described above, a protected or restored shellfish population may be viewed as an environmental asset that provides a stream of ecosystem services over time. An important factor for planners and decision makers to consider when weighing the benefits and costs of shellfish enhancement is the time frame over which the benefits and costs occur. When making decisions about changes to an ecosystem, a long time horizon is typically needed (Ludwig et al. 2005). For example, in discussing the role of the timing of benefits and costs related to the potential introduction of the Pacific oyster into Chesapeake Bay, Lipton (2008b) notes that it would take several years to restore oyster resources in Chesapeake Bay to a level that would support the level of a fishery. Since much of the restoration expense will occur early in the process, this timing will have an impact on calculation of net benefits.

Economic discounting is the process of weighting the sequence of costs or benefits over time (Ludwig et al. 2005). In general, individuals will discount values of things in the future in comparison to the same things in the present. To reflect this positive time preference, the standard economic approach is to use a discount rate to express a time series of effects as an equivalent present value. The level of the discount rate substantially affects the importance of services that occur far in the future—even

[23] As discussed in Section 3.2.2.3, it is possible that waterfront property owners in close proximity to large-scale shellfish cultivating operations may have negative values for this coastal zone use, related to such negative amenities as odors, noise, visual disturbance and other nuisances associated with such operations.

small differences in a discount rate for a long-term restoration project can result in order-of-magnitude differences to the present value of net benefits (National Research Council 2004).

Where the basis of comparison is strictly financial, economists feel confident using the opportunity cost of invested funds as the discount rate. However, for most ecosystem services, a financial opportunity cost is inappropriate and forces an emphasis on services that occur in the near future (National Research Council 2007b). Ludwig et al. (2005) argue that an appropriate consideration of economic and ecological uncertainties would lead planners and decision makers to discount benefits of ecosystem services at the lowest possible rate over long time horizons.

5.3 Limits of Economic Valuation

Apart from the technical problems with economic valuation methods discussed in Section 3.1, planners and decision makers should also be aware that there are criticisms of the basic principles underlying the application of these methods, especially with respect to valuing ecosystems. Some of these criticisms contend that economic valuation methods are inherently inadequate because they are based only on the preferences of the current generation and neglect the ethical issue of the intergenerational allocation of natural resource endowments. For example, Berrens et al. (1998) note that irreversible ecosystem losses involve intergenerational equity issues because they constrict the choice sets of future generations.

Other critics focus on the fact that economic valuation is rooted in anthropocentric or human-centered benefits, i.e., it rests on the basic assumption that value derives from what people find useful. Some would argue that human uses and the values to which they give rise are not deserving of any special consideration when it comes to a decision on whether or not to preserve an ecosystem or a certain species (National Research Council 2004). This non-anthropocentric or biocentric viewpoint assumes that all living things have value even if no human being thinks so. According to one interpretation of this notion of non-anthropocentric intrinsic value, non-human species have moral interests or rights unto themselves (National Research Council 2004).[24]

This reference to morals, rights and duties implies an ethic that rejects the assumption that humans even have a choice regarding whether or not to protect a species or ecosystem; rather, it is seen as an obligation (Mazzotta and Kline 1995; National Research Council 2004). These arguments are inconsistent with the economic principle of trade-offs between the provision of an ecosystem service and something else that can also be quantified by the dollar metric. Instead, they present individuals with the moral imperative that we ought to preserve all ecosystems and associated species (Mazzotta and Kline 1995). Brumbaugh and Toropova (2008) allude to these precepts when they note that some individuals around the U.S. who have initiated small-scale, community-based shellfish enhancement projects are motivated by the belief that the ecosystem services these projects provide have intrinsic value.[25]

As demonstrated by an increasing number of studies (e.g., Edwards 1986; Mazzotta and Kline 1995; Kotchen and Reiling 2000; Rekola et al. 2000; Spash 2000), the presence of such motivations can be a significant source of validity problems for economic valuation methods. As Costanza et al. (1997)

[24] The conceptual framework for The Millennium Ecosystem Assessment places human well-being as the central focus for assessment, but it recognizes that the actions people take that influence ecosystems result not just from concern about human well-being but also from considerations of the intrinsic value of species and ecosystems (The Millennium Ecosystem Assessment 2005).

[25] Flimlin (2008) succinctly summarizes the motivation question as follows: "… what, or who, has influenced these people to focus their careers, livelihoods, or hobbies toward small invertebrates locked within two opposing shells?"

and Pearce and Moran (1994) state, concerns about the preferences of future generations or ideas of intrinsic value translate the valuation of ecosystem services into a set of dimensions outside the realm of economics.

The issue of intrinsic value within economic valuation is currently unresolved (Hanley 2000). The National Research Council (2004) notes that this dilemma raises the question of what, if any, metric might be used to quantify, or at least rank, intrinsic values. According to the National Research Council, choices made by planners and decision makers are often made on the basis of information about many sources of value, including intrinsic and moral values, as well as economic values, and some decision rules seek to incorporate different types of values explicitly. As an example, the National Research Council describes decision rules that imply adherence to moral principles or a premise of intrinsic value unless the cost is too high; these decision rules incorporate concern about both intrinsic value and economic welfare, and implicitly allow some trade-offs between the two. Similar trade-offs are also implied by decision rules that apply a benefit-cost test to environmental policy choices but constrain the decisions to ensure that certain conditions reflecting intrinsic value are not violated (National Research Council 2004). Possible constraints include ensuring that levels of critical ecosystem services do not fall below standards necessary to ensure their continuation. In such cases, information about benefits and costs as determined by economic valuation will be a useful input into the policy decision but will not solely determine it (National Research Council 2004).

6 Economic Impact Analysis

Economic valuation and economic impact analysis are two widely used but distinctly different economic measures (TCW Economics 2008). As discussed in Section 3, economic valuation is a measure of the net economic welfare derived by society from policy or program changes. Economic impact analysis, on the other hand, provides planners and decision makers with information on how policy changes affect economic activity, as measured in terms of sales/output, value added, income, employment and tax revenues, in communities, counties, or even at the state or national level (TCW Economics 2008). Economic impact analysis is the focus of our Task 3 work.

Economic impacts are typically estimated using an input-output model—a methodology that models the linkages between input supplies, outputs and households in a regional economy that can be used to predict the impact of changes on economic activity within the region. Through the use of multipliers an input-output model provides measures of the total effects throughout the economy of a unit change in direct or initial spending, including indirect effects (businesses buying and selling to each other) and induced effects (household spending based on the income earned from the direct and indirect effects).

Several studies have measured the economic impacts of projects related to shellfish enhancement. Based on the results of an input-out model, it was estimated that oysters worth $1 million in dockside value in Chesapeake Bay generate an estimated $36.4 million in total sales, $21.8 million in income, and 932 person-years of employment (NOAA Fisheries Office of Habitat Conservation undated-a). Athearn (2008), Gardner Pinfold Consulting Economists Ltd. (2003) and O'Hara et al. (2003) estimated the economic impacts of shellfish aquaculture in Maine, while Frank Harmon Architect and Olympus Aquaculture Consulting (2008) projected the impacts of a hatchery in Virginia for the production of oyster larvae and spat. Philippakos et al. (2001) utilized an input-output methodology to estimate the direct, indirect and induced economic impacts of the cultured clam industry in Florida. Burrage et al. (1990) examined the regional economic impacts of a project intended to revitalize the northern Gulf Coast oyster industry by relaying oysters (moving oysters from leases under compromised water quality to leases in cleaner, approved waters before final harvest).

In Washington State, an early study by Bonacker and Cheney (1988) measured the direct economic impacts of shellfish culture in Willapa Bay. The study examined expenditure patterns of industry employees but did not calculate multiplier effects. According to a 1987 study of Washington's aquaculture industry conducted by the Washington State Department of Trade and Economic Development (Inveen 1987), the ratio of total jobs to direct jobs for the oyster industry was 1.17. That is to say, for every one job directly related to the industry, 0.17 additional indirect jobs were generated in other industries throughout the State. An economic impact analysis conducted in the early 1990s by Conway (1991) suggested that, on average, each job in Washington's oyster industry supported 1.13 additional jobs elsewhere in the state economy—this constitutes an employment multiplier for the oyster industry equal to 2.13. Wolf et al. (1987) of the Economic Development Council of Mason County estimated the economic impact of the County's oyster industry using the employment multiplier of 1.17 from the Washington State Department of Trade and Economic Development's 1987 study. The analysis was updated in 2002 using the same employment multiplier (Economic Development Council of Mason County 2002).

7 References

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