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Linking Ecological Risk Assessment and Economic Benefits: a Case Study of the Portsmouth Naval Shipyard Superfund Risk Assessment

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1. INTRODUCTION

1.1 Background

Ecological risk assessments are used to support an array of decisions across EPA programs, including, for example, setting national air quality standards, establishing site-specific waste clean-up goals, and specifying effluent guidelines for particular industries or limits for particular water bodies. For many of these decisions (depending on the statutory authority), EPA also has a mandate to assess the relative cost and benefits of proposed regulations to society. The current analysis of many rules is, however, sparse in the description of benefits related to ecosystem services and improved ecosystem functions. This is not surprising because describing the intrinsic worth of environmental services represents a significant challenge. In particular, to develop an improved ecological benefits analysis capability, methods must be developed to "translate" ecological assessment endpoints into descriptors that can be understood in terms of their societal value and benefit (whether or not such benefits can be explicitly monetized).

Ecological risk assessments may use a wide range of measures to characterize risks to organisms, populations, communities and ecosystems/ ecological functions. For example, ecologists have developed a suite of indices to measure community health and to measure the level of community impacts from stressors (such as species richness, diversity indices, and dominance by opportunistic species which is characteristic of a disturbed community). In some more detailed assessments, ecologists model changes in populations or, in some cases, changes in ecosystem composition (for example, using relative toxicities to various compartments of the ecosystem and/or using food chain models). Other, even more complex, systems dynamic models are also being developed that represent overall ecological resilience and sustainability.

However, in practice, needed data are often not available or systems are too complex to characterize risk at higher levels of biological organization; therefore, in regulatory contexts, ecological risks often have been characterized using hazard quotient methods that compare current or anticipated exposures to an ecologically relevant benchmark values (e.g., ambient water quality criteria).

While these assessments support the evaluation of risks to environmental values, the particular measures are often not directly or obviously linked to potential consequences for societal, or specifically economic, values. In other cases, the logical link is clear between the assessment endpoint and societal values (such as global biodiversity) but quantifying the link is difficult. Nonetheless, decision makers need ways to describe the value of protecting these environmental entities in order to make rational regulatory decisions.

1.2 Purpose of This Report

In November 2004, EPA issued a draft document entitled "NCEE Draft Ecological Benefits Assessment Strategy" (SAB Review version, November 11, 2004). That document outlines a series of action items to advance ecological benefits assessment within the EPA regulatory context. This report responds to particular actions identified in that report, specifically:

- "*Explore methods for expanding the use of ecological risk assessment information in economic benefits assessments*,"
- "*Create a catalogue of existing population models and develop guidance on model selection and use,*"

The efforts presented in this report will help develop ways to use available risk assessment techniques to describe impacts on ecosystems in terms that can be understood by decision makers and are useful for economic valuation.

This report builds upon the findings from *Linking Ecological Risk Assessment and Economic Benefits*, an earlier report on this topic (Abt Associates Inc., 2005). In that report, we reviewed ecological risk assessment techniques used by U.S. EPA and other government agencies, discussed the availability of literature on economic valuation that could be applied to the types of endpoints evaluated by each risk model, and conducted several ecological benefits assessment exercises based on published ecological risk studies.

In the current report, we conduct a more extensive economic evaluation of the results of one of the case studies from our previous report: the Portsmouth Naval Shipyard Superfund risk assessment (Johnston et al., 2002; U.S. Navy, 2000). We provide an overview of the ecological results of this risk assessment in Section 2 of this report. Then, in Section 3, we discuss the relationship between the ecological measures evaluated in the Naval Shipyard risk assessment and ecological services with economic value, identify challenges associated with trying to quantify that relationship, and then present the results of a valuation exercise that demonstrates how changes in ecological measures from the risk assessment can be used to predict changes in social welfare. Finally, in Section 4, we discuss future research needs and suggest steps that might contribute to a more unified approach to benefits analysis of ecological risks.

2. PORTSMOUTH NAVAL SHIPYARD ECOLOGICAL RISK ASSESSMENT

2.1 Background and Risk Assessment Methodology

The case study presented in this report is based on a Superfund risk assessment conducted by the U.S. Navy for the Portsmouth Naval Shipyard, located on Seavey Island, Maine, near the border with New Hampshire. Due to a long history of poor waste disposal practices at the Shipyard, a number of nearby areas were contaminated with pollutants such as metals, cyanide, polychlorinated biphenyls, phenols, oils, grease, sludge, solvents, asbestos, blasting grit, and incinerator ash. In 1994, the Shipyard was recognized as a Superfund site and was added to the National Priorities List. As part of the ensuing remediation process, the U.S. Navy conducted a detailed risk assessment to identify the potential ecological effects of past waste disposal practices at the site. The results of that risk assessment are presented in U.S. Navy (2000) and in Johnston et al. (2002).

The Superfund risk assessment framework used by U.S. Navy (2000) and Johnston et al. (2002) combines a variety of ecological indicators to evaluate the risk to six different ecological communities within the Naval Shipyard study area: the pelagic community, the epibenthic community, the infaunal benthic community, the eelgrass community, the salt marsh community, and the avian community. For each community, the authors present a variety of indicators of the evidence of ecological effect and exposure. The authors use a conceptual model to combine these indicator measures together to generate an overall indicator of the evidence of risk to each community.

The conceptual model used in the risk assessment utilizes a weight of evidence approach, which is recommended for ecological risk assessments. The weight of evidence approach takes into account the quality and strength of available data on each endpoint, for example, the quality of data obtained on winter flounder abundance and the strength of the link between winter flounder abundance and overall pelagic health. A wide range of data was collected because of the Superfund site status, which requires monitoring and assessment of ecological conditions. These data would not necessarily be available at all sites requiring risk assessment under the Superfund program. However, the methods used are standard approaches, and do not require intensive population or ecosystem modeling efforts, which not all risk assessors (such as state and local agencies) can readily conduct.

In line with EPA's recommended guidelines for ecological risk assessments, this Superfund ecological risk assessment evaluates risk to selected assessment endpoints, by measuring levels of exposure and effect, and comparing these levels with benchmark levels of toxicity from other studies, or to measurements from uncontaminated reference sites with similar ecological characteristics. Assessment endpoints were selected as the parts of the ecosystem that were to be protected, and these consisted of the different communities of importance in the estuarine system: the pelagic community, epibenthic community, infaunal benthic community, eelgrass community, salt marsh community, and the avian community. Although these communities as a whole are not easily tied to economic values, measures were taken for various receptor species, some of which may be more easily related to economic values. This approach allows for sampling of representative species within each community, and minimizes the time and labor that would be needed for sampling all possibly affected species. Species selected were either important to the ecology of the estuary, sensitive to the chemicals of concern, and/or are important for aesthetic, recreational, or commercial reasons.

Chemical disposal, landfills, disposal of waste oil, and other practices involved in repair of submarines at the shipyard released wastes containing metals, cyanide, polychlorinated biphenyls (PCBs), phenols, oils, and grease into the surrounding estuary. Tidal flats were also used as landfills, for sludge, solvents, asbestos, blasting grit, incinerator ash, waste oils, and dredge spoils. The assessment focused on areas that were likely to accumulate contaminants, and thereby pose a greater risk, because of patterns in water flow in the estuary. This process involved hydrological modeling to determine contaminant transport pathways. Six areas of concern were evaluated (see Figure 2-1), and chemicals of concern were chosen that were found above safe levels in the estuary and within receptor species, and were associated with discharge points from the shipyard. Chemicals of concern included: lead, mercury, copper, chromium, nickel, zinc, silver, arsenic, cadmium, polycyclic aromatic hydrocarbons (PAHs), PCBs, and the pesticide DDT (dichlorodiphenyl trichloroethane) and some of its metabolites. These chemicals have a wide range of effects depending upon the affected species, including disease, internal lesions, death, and impairment of growth, fertilization and development. These biological effects could also cause changes in behavior that may affect survival and reproduction. This situation is typical of many contaminated sites, where there are often a variety of chemicals to consider. However, chemical mixtures add uncertainty because of the possibility of cumulative effects that may occur, whereas the method of comparing individual contaminant levels to benchmark concentrations only considers effects of individual chemicals.

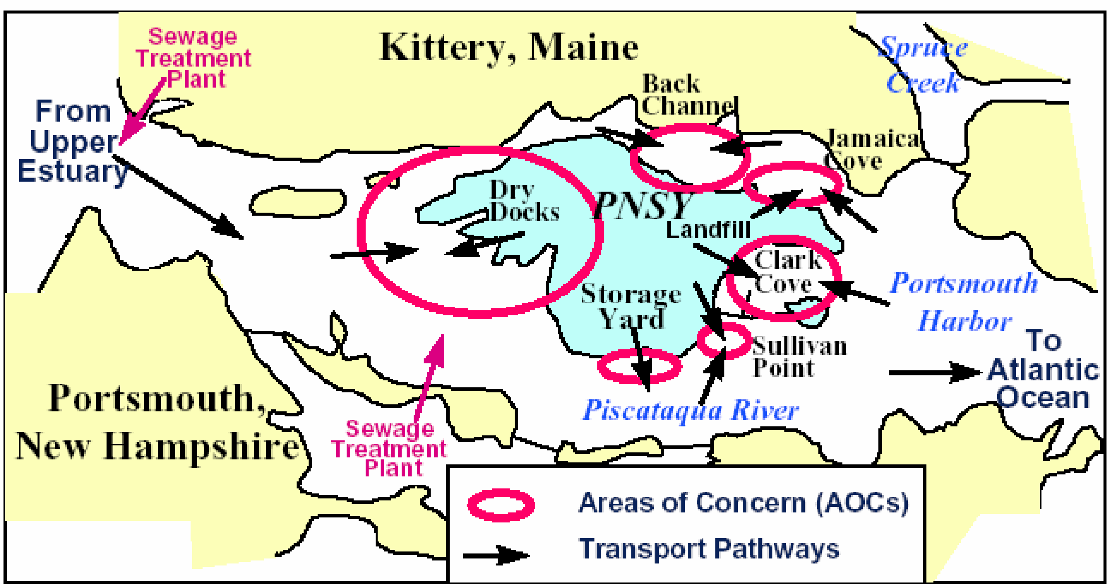

Figure 2-1. Modeled areas of concern (shown in red circles) based on patterns of water flow in the estuaries of Portsmouth Naval Shipyard. (Reproduced from U.S. Navy, 2000, p.1-9)

2.2 Ecological Endpoints Assessed

Pelagic community health

The pelagic community consists of diverse species in the open surface water of the Piscataqua River and the estuary, including phytoplankton, zooplankton, and pelagic fish. This habitat provides food for estuarine birds and supports commercial and recreational fisheries nearby. Many of the chemical stressors identified at the Shipyard could directly enter the pelagic system through direct discharge, surface water runoff, ground water discharge, soil erosion, and wind transport.

Receptor species measured in assessing risk to this community include phytoplankton, flounder, blue mussels, and sea urchins. Phytoplankton community biomass indicates the level of primary production, and it is an important food source for invertebrates and fish. Other species of phytoplankton have been known to experience reduced population growth rate in response to metals exposure. Blue mussel (*Mytilus edulis*) is an important source of food for birds, fish, starfish, and is consumed by people. Because they are sedentary, mussels can accumulate contaminants and thus provide information on contaminants in the system not detected in water quality tests. Toxicity to sea urchin larvae were studied because many species deposit their sperm and larvae in the water, and this serves as an indicator for larvae and sperm survival for these other species. Winter flounder (*Pleuronectes americanus*) is important ecologically and economically, as a food source for birds and other fish, and in recreational and commercial fishing. They feed upon benthic organisms, and thus represent a worst-case scenario because they may accumulate contaminants from bottom sediment as well as through benthic organisms and the water column. Contaminant effects previously found include impacted survival, reproduction, growth, and predator-prey interactions (p. 3-14).

Epibenthic community health

Exposure to this community could occur through the water column and also re-suspension of bottom sediment. The epibenthic community assemblage includes bivalves, crustaceans, echinoderms, and demersal fish. Lobster (*Homarus americanus*) was considered an indicator for other epibenthic species, because it can accumulate contaminants from the water column, sediment, and in the food chain, and is a food source for fish and humans. As they live relatively long and have primitive metabolic systems, and thus may not metabolize contaminants as well as fish and other invertebrates, their density and contaminant concentrations in tissue were measured as indicators for potential exposure for other epibenthic species. Fucoid algae (*Ascophyllum nodosum*) would be exposed to contaminants through the water column, and may accumulate chemicals because they live relatively long and cannot regulate their uptake. Algae biomass and tissue contaminant concentrations were assessed as measures of effect and exposure, respectively. Other types of macroalgae have been shown to experience impacts to their growth and reproduction from different pollutants. As mentioned previously, blue mussel, another receptor species, is an important food source in the estuary and has been known to respond to heavy metals and organic pollutants with decreased growth rate, physiological condition, and survival. They have also been shown to accumulate contaminants in proportion to the concentration in the water column (p.3-14). Their density, condition, and contaminant concentrations in their tissue were measured.

Benthic community health

The benthic community evaluated as part of the risk assessment includes infaunal species, including polychaetes and mollusks, which are primary consumers, and serve as food for fish, birds, and mammals. This community also supports commercial and recreational fishing for lobster, winter flounder and other fish. The infaunal species are potentially exposed to contaminants in the estuary through dissolved and particle-bound contaminants in sediment pore water. The contaminants of concern, particularly the heavy metals, remain associated with bottom sediment for long periods of time, thus increasing potential exposure in the benthic community. Potential impacts from various contaminants may include decreased survival, growth, and reproduction (p.3-13). Density and measures of richness and diversity were assessed for this community, as well as toxicity to amphipods.

Eelgrass community health

Eelgrass beds make up a large percentage of the estuary, and are depositional areas for suspended sediments because of the motion of the long grass blades of the eelgrass *Zostera marina*. In addition, the eelgrass could potentially accumulate contaminants through uptake through the sediment, and directly from the water column. This habitat harbors finfish and invertebrates such as lobster, and is an important feeding ground for aquatic birds. Eelgrass abundance, morphology, and contaminant concentrations were measured to assess eelgrass health.

Salt marsh community health

Salt marsh areas near the Shipyard were considered depositional areas for contaminated sediments, through water transport and also from direct discharge from nearby waste sites. Characteristic species include *Spartina alterniflora* (cord grass), *Spartina patens* (salt hay), *Distichlis spicata*, and *Juncus gerardii*. These grass species could potentially be exposed similarly to the eelgrass, through the water column and from the sediment. Many species are sheltered in the salt marsh habitat, including juvenile fish and minnows, birds, terrestrial animals, and invertebrates. It is also important in nutrient recycling and water filtering. Although these functions could not be measured directly, grass cover, morphology, and abundance and richness of invertebrate communities were measured (mollusks and amphipods) as indicators of the salt marsh community. Exposure to the community was also assessed through sediment and grass tissue concentrations.

Avian health

A variety of birds are found in the area of the Shipyard, including seabirds, diving birds, wading birds, shore birds, birds of prey, and salt marsh birds. These species would mainly exposed to contaminants in the estuary from ingesting contaminated prey or plants, but also through contaminated water or sediment, or through direct contact with contaminants. Possible effects include death, and reproductive and developmental impacts, and it is possible that various species may bioaccumulate some contaminants. However, as part of the risk assessment, measures of effect within bird species were not evaluated. Measures of exposure were evaluated, by determining levels of contaminants within food sources (prey and plant species), and comparing the maximum contaminant levels with dietary benchmark concentrations that are known to cause harm. Exposure through plants (eelgrass, salt marsh grasses, fucoid algae) was used to assess potential dietary exposure to Canadian geese and black ducks, and contaminant levels in flounder and mussels were used to assess dietary exposure for carnivorous ospreys, herring gulls, and omnivorous black ducks.

2.3 Ecological Risk Assessment Results

Table 2-1 summarizes findings from the risk assessment for Clark Cove, one of the most severely contaminated areas near the Naval Shipyard. For each broad ecological endpoint, the table presents evidence about risk based on different measures, and documents the risk assessment's conclusions about overall risk.

Table 2-1. Risk Assessment Results for Clark Cove

Ecological Assessment Endpoint	Evidence of Effect[a]	Evidence of Exposure[a]	Summary of Evidence	Magnitude of Risk	Confidence
Pelagic community health	Potential (M)	Low (M)	-Phytoplankton biomass within normal limits -Some contaminants higher in fish tissue compared to reference -Some evidence of flounder spleen pathology	Low	Medium
Epibenthic community health	No (M)	Elevated (M)	-Fucoid algae abundance within normal range -Juvenile lobsters abundant -Contaminant concentration in juvenile lobster tissue elevated -Mussel abundance within normal range -Some contaminants concentration elevated in mussel tissue	Low	Medium
Benthic community health	No (H)	Elevated (M)	-Benthic infaunal community within normal range	Low	High
Eelgrass community health	Potential (M)	Elevated (M)	-Eelgrass abundance/morphology within normal range -Eelgrass absent from one study site -Evidence of metal accumulation from sediment -Root biomass related to lead in sediment	Intermediate	Medium
Salt marsh community health	No (M)	Elevated (M)	-Some evidence of different morphology and lower abundance of salt marsh grasses in some sites -Salt marsh grass canopy height was related to sediment PAH concentration -Cover of vascular plants was	Low	Medium

Ecological Assessment Endpoint	Evidence of Effect[a]	Evidence of Exposure[a]	Summary of Evidence	Magnitude of Risk	Confidence
			greater at some sites		
Avian community health	*(No measures of effect studied)*	Negligible (M)	-Cumulative dietary exposure to contaminants through plants and prey species was negligible	Negligible	Medium

[a] Letters in parentheses refer to weight assigned to the endpoint in assessing magnitude of risk. Endpoint weight was based on data quality, strength of association, and study design (H=high, M=medium, L=low).
Source: Johnston et al. (2002).

3. ECONOMIC BENEFITS ASSESSMENT

This section of the report discusses the economic effects of the ecological changes identified in the Naval Shipyard risk assessment. Section 3.1 describes the economic benefit categories related to the ecological risk measures presented in the Naval Shipyard risk assessment. Section 3.2 discusses some challenges associated with creating quantitative linkages between ecological risk measures and those categories of economic benefits. Finally, Section 3.3 describes a valuation exercise that illustrates one potential method of valuing the ecological changes discussed in the Naval Shipyard risk assessment

3.1. Linkages Between Ecological Risk Measures and Economic Benefits

Table 3-1 lists the ecological measures of effect and exposure used in the risk assessment for the Naval Shipyard, as well as the assessment endpoints that these measures represent (U.S. Navy, 2000; Johnston et al., 2002). The table also describes economically valuable ecological services associated with each of those assessment endpoints and lists the types of economic values associated with those services. Because of the complex and interrelated nature of the components of the Naval Shipyard area ecosystems, it is impossible to list all potential linkages between ecological services and economic values. The ecological services and economic values presented in Table 3-1 should be viewed as a preliminary listing.

Table 3-1: Relationship Between Ecological Measures of Effect and Exposure, Ecological Services, and Economic Values at the Portsmouth Naval Shipyard Study Area

Ecological Measures of Effect and Exposure Used to Assess Potential Changes in Community Health[a]	Ecological Assessment Endpoint[a]	Related Ecological Services with Economic Values[b]	Economic Values
• Phytoplankton biomass • Mussel growth after 28 days • Sea urchin fertilization after exposure to water • Winter flounder abundance and size • Winter flounder spleen histopathology • Estuarine surface water concentration • Deployed mussel tissue concentration after 28 and 90 days • Seep water contaminant concentration • Winter flounder liver and fillet tissue concentration	Pelagic community health	• Habitat for recreationally and commercially valuable fish species	• Welfare gain from recreational fishing • Welfare gain from commercial fishing
		• Pelagic community health	• Non-use values for pelagic community health (as part of overall ecosystem health)
• Lobster density • Indigenous mussel density • Indigenous mussel shell length • Indigenous mussel condition index • Fucoid algae biomass	Epibenthic community health	• Habitat for recreationally and commercially valuable species (lobsters, mussels, etc)	• Welfare gain from recreational harvest of crustaceans and shellfish • Welfare gain from commercial harvest of crustaceans and shellfish

Ecological Measures of Effect and Exposure Used to Assess Potential Changes in Community Health[a]	Ecological Assessment Endpoint[a]	Related Ecological Services with Economic Values[b]	Economic Values
• Estuarine surface water concentration • Fucoid algae tissue concentration • Adule and juvenile lobster tissue concentration • Seep water concentration • Indigenous mussel tissue concentration		• Epibenthic community health	• Non-use values for epibenthic community health (as part of overall ecosystem health)
• Amphipod mortality after exposure to sediment • Benthic community richness • Benthic community density • Benthic community evenness • Concentration of acid volatile sulfide minus simultaneously extracted metal • Predicted pore water toxicity • Metal enrichment • Bulk sediment contaminant concentration	Benthic community health	• Benthic community health	• Non-use values for benthic community health (as part of overall ecosystem health)
• Eelgrass leaf morphology • Eelgrass root and rhizoe morphology • Eelgrass vegetative shoot density • Eelgrass reproductive shoot density; • Eelgrass ratio of leaves to shoots • Eelgrass spatial distribution • Bulk sediment contaminant concentration • Eelgrass leaf tissue concentration • Eelgrass root and rhizome concentration	Eelgrass community health	• Habitat and/or forage area for recreationally and commercially valuable fish and invertebrates (Atlantic cod, tomcod, winter flounder, cunner, rock crab, American lobster, etc) [1]	• Welfare gain from recreational fishing and harvest of crustaceans and shellfish • Welfare gain from commercial fishing and harvest of crustaceans and shellfish
		• Forage area for waterfowl[1]	• Welfare gain from bird watching and hunting • Non-use values for avian health
		• Sediment stabilization and prevention of shore and beach erosion[1]	• Reduction of damages and remediation costs associated with beach and shore erosion
		• Eelgrass community health	• Non-use values for eelgrass community health (as part of overall ecosystem health)
• Marsh grass cover • Marsh grass morphology • Amphipod abundance • Marsh grass leaf tissue concentration • Bulk sediment contaminant concentration	Salt marsh community health	• Habitat for invertebrates[2] • Spawning and nursery area for fish[2]	• Welfare gain from recreational fishing and harvest of crustaceans and shellfish • Welfare gain from commercial fishing and harvest of crustaceans and shellfish
		• Breeding and feeding habitat for wildlife[2]	• Welfare gain from bird and wildlife watching and hunting

Ecological Measures of Effect and Exposure Used to Assess Potential Changes in Community Health[a]	Ecological Assessment Endpoint[a]	Related Ecological Services with Economic Values[b]	Economic Values
		• Salt marsh community health[2]	• Non-use values for salt marsh community health (as part of overall ecosystem health)
• *No measures of effect were presented in Johnston et al. (2002)* • Dietary exposure to herbivore (Canada goose) • Dietary exposure to omnivore (black duck) • Dietary exposure to piscivore (osprey) • Dietary exposure to carnivore (herring gull)	Avian health	• Avian health	• Welfare gain from bird watching and hunting • Non-use values for avian health

[a] Ecological measures and ecological assessment endpoints taken from Navy (2000) and Johnston et al. (2002).
[b] Many of the ecological services listed in this column have little or no direct economic value, but may indirectly affect other ecological service flows that do have significant economic values. For example, although there is no direct human welfare gain from fish spawning in salt marsh habitat, there is a significant welfare gain from the effect of growth of fish populations on commercial and recreational angling.

References:
[1] *Wilbur (2004)*
[2] *Stratus Consulting (undated).*

3.2. Issues Associated with Linking Ecological Risk Measures to Economic Benefits

Linking ecological risk indicators to economic values is challenging. Although risk assessments typically focus on ecologically important measures of environmental effect and exposure, these measures are not necessarily relevant for estimating anthropocentric effects. In the case of the Naval Shipyard risk assessment, there are a variety of reasons why the ecological indicators and endpoints are challenging to value using economic techniques:

- Although some of the ecological endpoints used as inputs for the risk assessment model can be linked directly to economic values, other endpoints have only indirect relationships with economically valuable ecological service flows. For example, although winter flounder size and abundance has a direct impact on the welfare of recreational anglers, marsh grass morphology has at best a very indirect relationship with economic use values.
- The community level endpoints generated by the risk assessment are very general and thus difficult to use to evaluate effects on specific species with economic use values. For example, it would be very difficult to use an overall indicator of epibenthic community health to calculate changes to shellfish harvest.
- Many of the effect, exposure, and risk indicators are expressed qualitatively (particularly the broader ecosystem level indicators), and, even when quantitative, are not expressed as dose-response functions, making it difficult to use them to estimate specific economic values.
- The way that people value reductions in ecological risk, as opposed to ecological damages, is not well understood.

3.3. Valuation Exercise

Although calculating the economic effects of ecological changes is difficult, the results of such calculations can be very useful for analysts and decision-makers. As a step towards establishing an economic framework for evaluating such ecological changes, in the following pages we present a valuation exercise that illustrates one potential method of valuing the ecological changes discussed in the Naval Shipyard risk assessment. This exercise shows how changes in ecological endpoint measures can be used to estimate changes in the economic value of ecosystem services, even when the ecological endpoints are not direct measures of the ecosystem services with value.

The methodology that we used to value ecological changes at the Naval Shipyard study site can be summarized as follows:

1) Review the general ecological endpoints evaluated in the risk assessment and screen out all endpoints with low risks of exposure and effect (since the economic values associated with the ecological services provided by these endpoints are unlikely to be affected).
2) For each remaining endpoint:
    - Identify specific ecological measures that are likely to be related to the status of economically valuable ecological services provided by the ecosystem endpoint.
    - Use the observed changes in those ecological measures to estimate a range of possible changes in the ecological services with economic value (note that this step may require strong assumptions and a significant amount of professional judgment on the part of the researcher).
    - Use revealed or stated preference data from previous valuation research to estimate the range of welfare losses or gains associated with the estimated changes in the economically valuable ecological services.
3) Once all relevant categories of economic value have been evaluated, calculate the total economic value associated with all ecological changes by summing together all component economic values.

One key feature of this exercise is the calculation of *ranges* of possible ecological and economic changes. Estimating ranges of values helps to represent the substantial uncertainties inherent in both the ecological and economic steps of the valuation exercise. Additionally, an estimate of maximum possible damages may be of particular interest to decision-makers, particularly if the costs of remediation are expected to be large (and are known with a higher degree of precision). The maximum expected damages represent an upper bound on the benefits of remediation, and thus comparison of this worst-case environmental damages estimate with a more precise estimate of the cost of remediation gives an indication of the maximum net benefits (or minimum net costs) of a proposed remediation plan.

Another important feature of this exercise is explicit documentation of assumptions regarding the linkages between measured ecological endpoints and ecological services with economic value. Careful and transparent documentation of these linkages helps reviewers to understand the estimation methodology and to recognize the limitations associated with the resulting estimates.

Although we have taken steps to quantify the uncertainty inherent in this analysis, the results of this valuation exercise should still be interpreted with extreme caution. As discussed previously in Section 3.2, there are a number of reasons why it is difficult to use ecological indicators to estimate the economic value of ecological changes. In particular, we emphasize that many of the ecological endpoint measures we have as evidence of changes in economically relevant ecological services may not be good measures of those services. For example, in our analysis of recreational fishing, we use measures such as contaminant concentrations in flounder liver and spleen tissue as supporting evidence to infer that fish reproductive success may have been impaired and that fish populations may have decreased. The validity of this inference clearly depends on several significant assumptions regarding the relationships between contaminant levels, fish health, and fish populations.

The following sections document the results of our valuation exercise for the Naval Shipyard study area.

3.3.1 Screening Analysis

As discussed in Section 2, the risk assessment for the Naval Shipyard area evaluated six broad ecological assessment endpoints: pelagic community health, epibenthic community health, benthic community health, eelgrass community health, salt marsh community health, and avian health. We began our analysis by screening the risk assessment results for each of these endpoints. Economic services provided by an ecosystem are unlikely to change if the ecosystem itself is unaffected; thus, our analysis excluded ecological endpoints for which the majority of measures of effect and exposure indicated high confidence of low or negligible risk.

Table 3-2: Rationale for Selection of Ecological Endpoints for Valuation Exercise

Ecological Assessment Endpoint	Risk Assessment Results	Rationale for Inclusion or Exclusion
Pelagic community health	Low risk (medium confidence)	Include, based on medium confidence of low risk.
Epibenthic community health	Low risk (medium confidence)	Include, based on medium confidence of low risk.
Benthic community health	Low risk (high confidence)	Exclude, based on high confidence of low risk.
Eelgrass community health	Elevated risk (medium confidence)	Include, based on elevated risk.
Salt marsh community health	Low risk (medium confidence)	Include, based on medium confidence of low risk.
Avian health	Negligible risk (medium confidence)	Exclude, based on negligible risk.

After screening out obviously unaffected ecological endpoints, we were left with a collection of ecological endpoints for which there was some possibility of harm. By combining information about the risk of harm for each ecological endpoint (from the proceeding table) with the list of economic services provided by each endpoint (from Table 3-1), we were able to identify the set of economically valuable ecological services provided by ecosystems in the Naval Shipyard study area that might potentially be affected. We used this information to screen out categories of economic value that were unlikely to be affected by environmental changes at the Naval

Shipyard study site. Table 3-3 presents the results of this analysis. For each category of economic value, the table lists relevant ecological endpoints and documents whether or not we chose to include each category of economic value in the final valuation exercise. The table shows that in some cases we based our valuation decision on other factors, such as our goal of valuing a range of different ecological systems.

Table 3-3: Rationale for Selection of Categories of Economic Value for Valuation Exercise

Economic Value Related to Ecological Service	Relevant Ecological Assessment Endpoints[a]	Valuation Decision and Rationale
Welfare gain from recreational fishing and harvest of crustaceans and shellfish	• Pelagic community health • Epibenthic community health • Eelgrass community health • Salt marsh community health	Include. A number of the ecosystems that support fish populations (and thus influence recreational fishing) have low or elevated risk of harm.
Welfare gain from commercial fishing and harvest of crustaceans and shellfish	• Pelagic community health • Epibenthic community health • Eelgrass community health • Salt marsh community health	Exclude. The risks to commercial fishing are likely to be similar to the risks to recreational fishing, so we decided to evaluate only one of these categories of value.
Welfare gain from bird watching and hunting	• Eelgrass community health • Salt marsh community health	Include. We evaluated this category of benefits as part of our analysis of the total value of salt marsh community health.
Welfare gain from wildlife watching and hunting	• Salt marsh community health	Include. We evaluated this category of benefits as part of our analysis of the total value of salt marsh community health.
Reduction of damages and remediation costs associated with beach and shore erosion	• Eelgrass community health	Exclude. Although there is an elevated risk of harm to eelgrass habitat, we chose not to evaluate this category of benefits because of time and resource constraints.
Non-use values for pelagic community health (as part of overall ecosystem health)	• Pelagic community health	Include. We evaluated non-use values for pelagic community health as part of our analysis of non-use values for water quality.
Non-use values for epibenthic community health (as part of overall ecosystem health)	• Epibenthic community health	Include. We evaluated non-use values for epibenthic community health as part of our analysis of non-use values for water quality.
Non-use values for eelgrass community health (as part of overall ecosystem health)	• Eelgrass community health	Exclude. Although there is an elevated risk of harm to eelgrass habitat, we chose not to evaluate this category of benefits because of time and resource constraints.
Non-use values for salt marsh community health (as part of overall ecosystem health)	• Salt marsh community health	Include. We evaluated this category of benefits as part of our analysis of the total use and non-use values associated with salt marsh community health.
Non-use values for avian health	• Eelgrass community health • Salt marsh community health	Include. We evaluated this category of benefits as part of our analysis of the total use and non-use values associated with salt marsh community health.

After consideration of the various categories of economic value that might be affected by changes in the ecological services provided by affected communities, we decided to evaluate economic effects on use values from recreational fishing, non-use values for water quality[1], and total values for salt marsh services. The methodology that we used to estimate these three categories of value is described in the following sections.

3.3.2 Calculation of Selected Economic Values Associated with Ecological Endpoints

3.3.2.1 Recreational Angling

We began the recreational angling analysis by identifying ecological endpoints that would be useful as inputs to a benefit transfer analysis based on existing economic valuation studies. Since most past economic studies of recreational fishing have focused on evaluating willingness-to-pay (WTP) for changes in catch rates, we focused on ecological measures that could be used to estimate changes in catch rates. Catch rates are determined by a variety of environmental factors, including the size of fish populations, the vigor and healthiness of individual fish, water temperatures, and the presence or absence of forage species. Of these measures, we chose fish population size as the single measure with the strongest influence on catch rates. We then attempted to estimate the economic value of changes in recreational angling by (1) identifying measures of ecological effect and exposure from the Naval Shipyard risk assessment that are likely to be correlated with fish population size, (2) using those measures to estimate the percentage and absolute change in fish populations in the study area, (3) using the local change in fish populations to estimate the change in local recreational catch, and (4) using a benefit transfer approach to evaluate the economic value of the change in catch.

We identified six measures of ecological effect and exposure from the Naval Shipyard risk assessment—contaminant concentrations in sediment, winter flounder size, winter flounder spleen histopathology, winter flounder liver tissue concentration, winter flounder fillet tissue concentration, and sea urchin fertilization test results—that are likely to be indicators of toxic effects on fish, particularly reproductive effects, and thus ultimately affect fish population levels. Table 3-4 presents Naval Shipyard risk assessment data for each measure. The table also shows the hypothesized relationship between each endpoint measure and populations of recreational fish species. Finally, the table presents our estimate of the change in recreational fish populations that can be inferred from each ecological measure, as well as an explanation for our estimate.

[1] Note that non-use values for water quality, as defined in this report, include non-use values for aquatic life (such as fish populations).

Table 3-4: Ecological Endpoint Measures Related to Population Levels of Recreational Fish Species

Measure of Ecological Effect or Exposure[a]	Data and Results from Risk Assessment[b]	Relationship to Populations of Recreational Fish Species	Quantitative Estimate of Effects on Recreational Fish Species[c]	
			Change in Population	Explanation
Quantitative Evidence				
None of the evidence presented in the risk assessment was sufficient to use as a basis for a quantitative estimate of changes in fish populations.				
Other Supporting Evidence				
Sediment concentrations	Fluorene and anthracene exceed the Effects Range-Low (ER-L) for marine sediments. Average concentration of fluorene was 64.9ng/g compared to the ER-L of 19 ng/g (hazard quotient of 3.4); average concentration of anthracene was 219.5 ng/g compared to the ER-L of 19 (hazard quotient of 2.6).	ER-L levels for various chemicals have been established as a lower-bound concentration in marine sediment that causes effect to a range of aquatic biota. Sediment concentrations exceeding these benchmarks indicate an increased chance of health and reproductive effects to species in the system, including fish. Protective concentrations established for water indicate that reduced reproductive success can occur with concentrations of PAH's as low as 8mg/L (British Columbia Ministry of the Environment, 1993). As the concentrations of fluorene and anthracene exceed this concentration in sediment, it is likely they could also result in lowered reproductive success.	Sediment concentrations of organics above the ER-L have been associated with 40 to 60% increased chance of effects (NOAA, 1999). However, this refers to the particular effect observed (e.g., the suppression of egg production), rather than the resulting effect on population.	
Winter flounder liver tissue concentration	Fluorene and anthracene were higher in liver tissue of fish in Portsmouth Harbor than fish from reference site (IX.1-3). Fluorene concentration was 50% greater at Portsmouth Harbor; anthracene was 250% greater (VI.62).	High liver tissue concentrations may be associated with reduced fish population health. Increased exposure to polycyclic aromatic hydrocarbons can also cause reproductive effects in fish (Johnson et al., 2002).	These results indicate that increased exposure to PAHs has occurred in fish, and that fish in the estuary have accumulated these chemicals in their tissues. PAHs in general have been associated with reproductive effects, such as lower egg production, and lower hatching rates, which hypothetically could in turn cause population changes.	
Winter flounder size	Slightly larger in Portsmouth Harbor than in reference site (6-8). However, data is not sufficient to use to draw conclusions (IX.1-3).	The presence of larger fish may be indicative of a healthier fish population. However, it may also indicate that few young fish are surviving—either because of low recruitment or because of selective effects of pollution on younger fish.	Increase in flounder size could be indicative of either positive or negative changes in fish populations.	

Winter flounder spleen histopathology	Abnormal spleen pathology observed in Portsmouth Harbor *and* reference site; condition not attributable to local causes. (IX.1-3)	Abnormal spleen pathology may be associated with reduced fish population health.	No difference between study site and reference site. While abnormality is observed, it is observed at both Portsmouth Harbor and the reference site, so it cannot be attributed to the contamination at Portsmouth Harbor.
Winter flounder fillet tissue concentration	Zinc was higher in tissue of fish in Portsmouth Harbor than fish from reference site (IX.1-3).	Higher fillet tissue concentrations may be associated with reduced fish population health.	The significance of this measure is not clear from the risk assessment; however, exposure to zinc can have lethal effects on fish (Woodling et al., 2002).
Sea urchin fertilization	Fertilization was partially inhibited when eggs were exposed to water samples from one of the study sites, but the magnitude of the effect was low (IX.1-2).	Depressed sea urchin egg fertilization rates may be indicative of the effects of contaminants.	These results are not directly applicable to fish populations, but indicate that impacts on reproductive success may occur in the study area for aquatic life.

[a] The Naval Shipyard risk assessment also evaluated winter flounder abundance, but since it provided no data on this measure, we did not include it in our calculations.
[b] References in parentheses refer to section and page numbers in the Naval Shipyard risk assessment report.
[c] Note that the percentage changes described in this column are best estimates of the entire change in recreational fish populations, based on each measure. Thus, these percentages changes are not cumulative across measures. The total change in recreational fish populations should be calculated using the average of these percentage changes.

Source: U.S. Navy (2000).

As shown in Table 3-4, none of the ecological measures presented in the Naval Shipyard risk assessment were sufficiently relevant to use to estimate changes in fish populations at the study site. However, because the sediments exceed the ER-L for PAHs, and because these compounds have been associated with reproductive effects, it is reasonable to assume that there may be some effects on fish populations. Further, the supporting evidence indicates that fish populations have been exposed to these compounds, and that depression of fertility also occurred in sea urchins exposed to water samples from the study site. Ideally, potential effects on reproductive outcomes could be used to estimate a change in the parameters of a matrix or other population dynamics model, which would then yield an estimate of the change in populations that may occur in recreational fish. Although we lack the information to implement such an approach, for the purpose of illustration we evaluate four population changes: 0 percent, -5 percent, -15 percent, and -40 percent. Since much of the supporting evidence from Table 3-4 indicates that effects on fish are likely to be relatively minor, we believe that the actual change in fish populations is likely to occur in the low end of this range, if at all.

Based on our estimate that total recreational catch of all species in the study area was 46,698 fish prior to contamination, we estimated that changes of 0 percent, -5 percent, -15 percent, and -40 percent would correspond to population changes of 0 fish, -2,335 fish, -7,005 fish, and –18,679 fish, respectively.[2] This calculation is predicated on the assumption that changes in catch rates are directly proportional to changes in fish populations. Table 3-5 shows the details of this calculation.

[2] Refer to Appendix A for documentation of our estimate of total recreational catch at the Naval Shipyard study area.

Table 3-5: Estimated Change in Recreational Catch Near Naval Shipyard Study Site

Scenario	Baseline Catch (Number of Fish)	Percentage Change in Population	Change in Catch (Number of Fish)
Lower bound	46,698	-40	-18,679
Low middle	46,698	-15	-7,005
High middle	46,698	-5	-2,335
Upper bound	46,698	0	0

We monetized this estimated change in recreational catch using results from a recent meta-analysis of recreational valuation studies (Johnston et al., 2005a). The meta-analysis estimates the relationship between WTP to catch an additional fish, and resource, demographic, and study methodological characteristics. Table 3-6 presents the meta-analysis variables, the regression coefficients, and the input values assigned to each variable for each of four general species groups found in the study area. The following bullets explain how these values were assigned:

- The study methodology variables were set to values that reflect a nested RUM study conducted in the year 2000 with a high resulting response rate.
- *Age42_down*, *age43_up*, *trips19_down*, and *trips20_up* were set to values that reflect the average values in the survey dataset. *Income* was set to $46,393, the average of the median household income for Maine and New Hampshire from 2001 to 2003 (U.S. Census Bureau, 2005a).
- The species dummy variables were all set to zero, except *small_game_atl*, *flatfish_atl*, and *other_saltwater*, which were set to 1.
- *Cr_nonyear* was set to values that reflect species-specific average per-day catch rates for North Atlantic anglers (NMFS, 2003).
- *Shore* was set to reflect the average percentage of shore anglers in the North Atlantic region (NMFS, 2003).

Based on these values for the input variables, we calculated WTP per additional fish to be $6.93 for small game, $6.96 for flatfish, and $3.48 for other saltwater species. For further documentation of the meta-analysis results and variables, refer to Johnston et al. (2005a).

Table 3-6: Recreational Meta-analysis Regression and Predicted WTP per Fish (2003$)

Variable	Coefficient	Input Value for Variable		
		Small Game	Flatfish	Other Saltwater Species
Intercept	-1.4568	1	1	1
SP_conjoint	-1.1672	0	0	0
SP_dichot	-0.9958	0	0	0
TC_individual	1.1091	0	0	0
TC_zonal	2.0480	0	0	0
RUM_nest	1.3324	1	1	1
RUM_nonnest	1.7892	0	0	0
sp_year	0.0875	0	0	0
tc_year	-0.0397	0	0	0
RUM_year	-0.0029	24	24	24
sp_mail	0.5440	0	0	0
sp_phone	1.0859	0	0	0
high_resp_rate	-0.6539	1	1	1
inc_thou	0.0039	46.3925	46.3925	46.3925
age42_down	0.9206	0.2639	0.2639	0.2639
age43_up	1.2221	0.7361	0.7361	0.7361
trips19_down	0.8392	0.2472	0.2472	0.2472
trips20_up	-1.0112	0.7528	0.7528	0.7528
nonlocal	3.2355	0	0	0
big_game_pac	2.2530	0	0	0
big_game_natl	1.5323	0	0	0
big_game_satl	2.3821	0	0	0
small_game_pac	1.6227	0	0	0
small_game_atl	1.4099	1	0	0
flatfish_pac	1.8909	0	0	0
flatfish_atl	1.3797	0	1	0
other_sw	0.7339	0	0	1
musky	3.8671	0	0	0
pike_walleye	1.0412	0	0	0
bass_fw	1.7780	0	0	0
trout_GL	1.8723	0	0	0
trout_nonGL	0.8632	0	0	0
salmon_pacific	2.3570	0	0	0
salmon_atl_morey	5.2689	0	0	0
salmon_GL	2.2135	0	0	0
steelhead_pac	2.1904	0	0	0
steelhead_GL	2.3393	0	0	0
cr_nonyear	-0.0814	0.65	0.24	0.82
cr_year	-0.0521	0	0	0
catch_year	1.2693	0	0	0
spec_cr	0.6862	1	1	1
shore	-0.1129	0.24	0.24	0.24
errorvariance	0.6581	0	0	0
WTP per Additional Fish (2003$)		**$6.93**	**$6.96**	**$3.48**
Source: Johnston et al. (2005a)				

Table 3-7 shows how we combined the per-fish WTP values from Table 3-6 with our estimates of the change in recreational catch at the Naval Shipyard site. By multiplying the change in catch by WTP per fish for each species, we were able to calculate the total change welfare attributable to changes in recreational fishing quality at the study site. The total annual value of the estimated change in catch ranges from a lower bound of -$94,315 to an upper bound of $0. However, even for the most negative percentage change in fish populations evaluated (-40%), the associated economic loss is relatively small, especially in relation to the value associated with the change in water quality at the Naval Shipyards study site (discussed in Section 3.3.2.2, below).

Table 3-7: Annual Recreational Fishing Benefits, by Species

Species Group	**Change in Recreational Catch at Study Site**				**WTP per Fish (2003$)**	**Value of Change in Catch (2003$)**			
	Lower Bound	**Low Middle**	**High Middle**	**Upper Bound**		**Lower Bound**	**Low Middle**	**High Middle**	**Upper Bound**
Small Game	-8,223	-3,083	-1,028	0	$6.93	-$57,012	-$21,379	-$7,126	$0
Flatfish	-268	-101	-34	0	$6.96	-$1,864	-$699	-$233	$0
Other Saltwater	-10,189	-3,821	-1,274	0	$3.48	-$35,439	-$13,290	-$4,430	$0
Total, All Species	**-18,679**	**-7,005**	**-2,335**	**0**	**n/a**	**-$94,315**	**-$35,368**	**-$11,789**	**$0**

The results of this analysis are subject to a number of limitations and uncertainties. First of all, since we were unable to use the ecological measures from the case study to predict a range of changes in fish populations, the numerical examples presented in this section are hypothetical at best. In view of the supporting evidence presented in Table 3-4, it seems likely that the actual change in fish populations at the study site would fall within the range of values evaluated above. However, without more extensive modeling of the toxicological effects of observed pollutant concentrations on fish populations, it is not possible to estimate the actual change in fish populations. Most of the measures presented in Table 3-4 are at best indirect measures of factors that affect fish populations. Thus, to use changes in these measures to estimate fish populations would ignore potentially complicated relationships that could exist within the pelagic ecosystem.

Additional uncertainty is associated with the definition of the affected geographic area. The baseline estimate of recreational catch is derived from a simple “back-of-the-envelope” calculation in which statewide recreational catch is assigned to particular areas based on the number of linear miles of coastline in those areas. This calculation is based on the assumption that all geographic areas are equally productive recreational fishing areas—an assumption that is not likely to be true. Furthermore, because of the mobile nature of fish populations, environmental damages at the Naval Shipyard site may harm recreational fishing in other geographic areas.

In addition to these specific limitations, all of the general limitations and uncertainties discussed in Section 3.1 also apply to the recreational fishing analysis results.

3.3.2.2 Water Quality

The results of our screening analysis indicated that water quality in the Naval Shipyard area might have been affected by contamination at the site. Lower water quality may affect local residents who make use of the area's water resources for recreational activities. It may also affect local residents who do not make use of the site or its resources, if these individuals value the satisfaction of knowing that local water resources are in good condition and will be available for future generations. For this valuation exercise, we estimated the change in the economic value of this second type of benefit, commonly termed non-use value, by (1) identifying measures of ecological effect and exposure from the Naval Shipyard risk assessment that are likely to be correlated with water quality, (2) using those measures to estimate the change in water quality in the study area, and (3) using a benefit transfer approach to evaluate the non-use value of the change in water quality.

To measure water quality, we used the Resources For the Future (RFF) water quality ladder. The RFF water quality ladder is a ten-point scale linked to specific pollutant levels which, in turn, are linked to presence of aquatic species and recreational uses. Thus, a WQL of 10 indicates drinkable water, a WQL of 7 indicates that a water body is safe for swimming, and a WQL of 5 indicates that a water body supports game fish. We estimated that baseline water quality at the Naval Shipyards site was 7.5 on the water quality ladder, equivalent to a level slightly cleaner than necessary for swimming.

To estimate the change in water quality that occurred after contamination of the study site, we evaluated quantitative information from the risk assessment on the presence of contaminants in water, as well as a variety of other supporting measures. Table 3-8 presents Naval Shipyard risk assessment data for these measures, as well as the hypothesized relationship between each endpoint measure and water quality. Finally, the table presents our estimate of the change in water quality that can be inferred from the data, as well as an explanation for our estimate.

Table 3-8: Ecological Endpoint Measures Associated with Water Quality

Measure of Ecological Effect or Exposure	Data and Results from Risk Assessment[a]	Relationship to Water Quality	Estimate of Effects on Water Quality	
			Change in Water Quality	**Explanation**
Quantitative Evidence				
Contaminant presence in estuarine surface water and seep water	Contaminant concentrations in estuarine surface water were below ambient water quality criteria (AQWC) and were similar to reference areas. However, average levels of Cu (6.17 μg/L), Ni (8.77 μg/L), Pb (4.73 μg/L), and Zn (125.2 μg/L) were higher than their AQWCs in seep water.	Medium/low strength of relationship. Presence of contaminants in surface water is a good indicator of water quality, but presence in seep water is only a weak measure.	Reduction to WQL of 5.5 to 7 (cumulative chronic effects of exposure are not sufficient to make water unfit for game fish, but may make it less suitable for some highly sensitive species. Since seep water metal concentrations are higher than overall estuarine metal concentrations, this is a worst-case scenario).	Exposure to estuarine surface water is not likely to cause harm. However, exposure to seep water may have negative effects. Potential effects of exposure to metals in seep water are as follows: **Cu:** Exposure to 4.8 μg/L is two day EC50 for blue mussel embryos; exposure to 56 μg/L is 13 day LC50 for lobster (VII-3). Thus, exposure level at seeps (6.2 μg/L) is likely to have both acute and chronic effects. **Ni:** Exposure to 10 μg/L reduces ATP production in blue mussels; exposure to 17 μg/L reduces chlorophyll-a by 65% in diatoms after 2 days (VII-9). Thus, exposure level at seeps (8.8 μg/L) may have some minor chronic effects. **Pb:** Exposure to 10 μg/L stops diatom growth after 12 days; exposure to 21 μg/L reduces natural phytoplankton populations after 4 days and prevents sexual reproduction in red algae (VII-2). Thus, exposure level at seeps (4.7 μg/L) may have chronic effects. **Zn:** Exposure to 130 μg/L is 17 day LC-50 for lobster; exposure to 96-314 μg/L inhibits blue mussel embryo development by 50% after 3 days (VII-6). Thus, exposure level at seeps (125 μg/L) may have chronic effects.
Other Supporting Evidence				
Phytoplankton biomass	Mean chlorophyll *a* concentrations were low at study site compared to reference site, but still within "normal range" (IX.1-1).	The presence of phytoplankton may indicate a certain minimum level of water quality. However, significant natural variation in phytoplankton is expected.	Observed effects may be the result of reduced water quality, but the causal link is tenuous.	
Sea urchin fertilization	Fertilization was partially inhibited when eggs were exposed to water samples from one of the study sites, but the magnitude of the effect was low (IX.1-2).	Depressed sea urchin egg fertilization rates may be indicative of the presence of contaminants in water.	Indicates a small decline in water quality.	

Measure of Ecological Effect or Exposure	Data and Results from Risk Assessment[a]	Relationship to Water Quality	Estimate of Effects on Water Quality	
			Change in Water Quality	Explanation
Deployed mussel condition	No negative effects observed.	Inhibited deployed mussel growth would be indicative of sediment contamination and/or poor water quality.	Not indicative of an effect on water quality.	
Pb concentration in deployed and indigenous mussels	Average Pb concentration in mussels increased by 65% after deployment for 90 days, from 1.8 µg/g to 2.96 µg/g dry weight.	High strength of relationship. Mussels may bioaccumulate contaminants such as Pb (IX.1-2).	The fact that Pb concentrations increased by 65% indicates that either water or sediment is contaminated with lead.	
Flounder liver histopathology	Elevated levels of fluorine and anthracene were observed in flounder liver tissue.	Medium/low strength of relationship. Presence of contaminants may be indicative of lower water quality, but flounder are highly mobile and thus results are difficult to attribute to the study site.	May be indicative of reduced water quality, but effect is highly uncertain.	
[a] References in parentheses refer to section and page numbers in the Naval Shipyard risk assessment report.				
Source: U.S. Navy (2000).				

The table shows that lower and upper bound water quality levels after contamination are 5.5 and 7.0, respectively. Compared to the baseline water quality level of 7.5, these new levels represent a change of -2.0 and -0.5 on the water quality ladder. Since much of the evidence presented in Table 3-8 indicates that water quality effects are likely to be small, we believe that the change in water quality is likely to be closer to -0.5 than to -2.0.

We monetized these estimated changes in water quality using a three-step benefit transfer methodology:

- First, we estimated the number of households that might reasonably hold non-use values for water quality at the Naval Shipyard study area.
- Second, we used an aquatic resource meta-analysis benefits model to predict how much those households would be willing to pay to prevent the estimated water quality changes.
- Finally, we multiplied that number of affected households by average WTP for the water quality changes.

We assumed that the population likely to hold non-use values for water quality at the Naval Shipyard study area would include all households within 25 miles of the area. Using U.S. Census Bureau data, we estimated this population to include 155,640 households.[3]

[3] Refer to Appendix B-2 for documentation of this calculation.

To estimate how much each household would be willing to pay to avoid the maximum water quality change at the reservoir, we used a meta-analysis model developed during EPA's analysis of the benefits of the final section 316(b) rule for Phase II facilities (Johnston et al., 2005b). This model analyzes the relationship between resource, demographic, and methodological characteristics and total annual WTP for improvements to surface water quality and aquatic habitat. The meta-analysis expresses improvements in water and habitat quality using the RFF water quality ladder.

Table 3-9 presents the meta-analysis variables and coefficients, as well as the values assigned to each variable. For detailed information about the variable definitions, refer to U.S. EPA (2004). The following bullets briefly explain how we assigned values to each of the variables:

- Selected survey methodological variables (*interview*, *year_indx*, *nonparam*, *discrete_ch*, *protest_bids*, *outlier_bids*) were assigned values that reflect a discrete choice survey format conducted through in-person interviews, with protest bids and outlier bids eliminated. Other survey methodological variables were set to zero.
- *Income* was set to $46,393, the average of the median household income for Maine and New Hampshire from 2001 to 2003 (U.S. Census Bureau, 2005a).
- The regional dummies were all set to zero, reflecting a change that takes place in the northeast.
- All of the resource description variables were set to zero, reflecting the default case of an estuary.
- The water quality variables *WQ_non* and *baseline* were assigned values that reflect a 2.0 point or 0.5 point decrease on the water quality ladder, from a baseline of 7.5 points, to an end level of 5.5 points or 7.0 points. The remainder of the WQ variables were set to zero, since all aspects of water quality may be affected.
- The variable *non-users* was set to one so that the model would predict total WTP for non-users only—which by definition includes only non-use values. This value is assumed to be a lower bound for the non-use value for users of the resource, who, being more familiar with the resource, might be expected to hold greater non-use values for it (independently of any use values they also hold).

Based on the variable assignments shown in the table, non-user WTP for changes of -2.0 and -0.5 on the water quality ladder is $13.77 and $5.63 per household, respectively. As a sensitivity analysis, we also calculated WTP based on a mail survey format (instead of an interview format). Under this alternative assumption, non-user WTP decreases by 53 percent, to $6.45 and $2.64, respectively.

Table 3-9: Estimating Non-use WTP for Changes in Water Quality

		Assigned Value[a]			
		WQL Change = -2.0		WQL Change = -0.5	
Variable	**Coefficient**	**Interview Survey Format**	**Mail Survey Format**	**Interview Survey Format**	**Mail Survey Format**
Intercept	6.0158	1	1	1	1
year_indx	-0.1072	31	31	31	31

Variable	Coefficient	Assigned Value[a]			
		WQL Change = -2.0		WQL Change = -0.5	
		Interview Survey Format	Mail Survey Format	Interview Survey Format	Mail Survey Format
discrete_ch	0.3956	1	1	1	1
Voluntary	-1.633	0	0	0	0
Nonparam	-0.4472	1	1	1	1
Income	0.00000058	46392.5	46392.5	46392.5	46392.5
wq_ladder	-0.3799	1	1	1	1
protest_bids	0.9537	1	1	1	1
outlier_bids	-0.8764	1	1	1	1
hi_response	-0.8094	1	1	1	1
single_river	-0.3378	0	0	0	0
single_lake	0.3193	0	0	0	0
multiple_river	-1.605	0	0	0	0
salt_ponds	0.7574	0	0	0	0
num_riv_pond	0.0791	0	0	0	0
regional_fresh	-0.0073	0	0	0	0
Southeast	1.1482	0	0	0	0
Plains	-0.8153	0	0	0	0
pacif_mount	-0.3125	0	0	0	0
multi_reg	0.5951	0	0	0	0
Nonusers	-0.5017	1	1	1	1
WQ_fish	0.2055	0	0	0	0
WQ_shell	0.2561	0	0	0	0
WQ_many	0.2332	0	0	0	0
WQ_non	0.4695	2.0	2.0	0.5	0.5
Fishplus	0.8052	0	0	0	0
Baseline	-0.1265	5.5	5.5	7.0	7.0
Interview	1.3252	1	0	1	0
Mail	0.5666	0	1	0	1
lump_sum	0.5954	0	0	0	0
nonfish_uses	-0.1412	0	0	0	0
median_WTP	0.2206	0	0	0	0
WTP per Household (2003$)		**$13.77**	**$6.45**	**$5.63**	**$2.64**

[a] Note that the average high and upper bound water quality changes are zero; thus, WTP for these changes is zero by default, and it is not necessary to use the meta-analysis regression equation to predict WTP.
Source: Johnston et al. (2005b); U.S. EPA (2004).

The final step in the calculation of non-use value for the water quality change at the Navy Shipyard site was to combine the estimate of the affected population with our estimates of household WTP. Table 3-10 shows the results of these calculations. Based on a total affected population of 155,640 local households, we estimated that the total annual non-use value of the water quality change has lower and upper bounds of -$2.1 million and -$0.4 million.

Table 3-10: Non-use Value Associated with Water Quality Changes

Scenario	Change in Water Quality (WQL)	Non-use Value per Household (2003$)		Number of Affected Households	Total Non-use Benefits (2003$)	
		Low	High		Low	High
Lower bound	-2.0	$5.63	$13.77	155,640	-$1,003,855	-$2,143,517
Upper bound	-0.5	$2.64	$6.45	155,640	-$410,593	-$876,734

There are several significant limitations and uncertainties associated with using the meta-analysis model to estimate non-use values for individuals affected by water quality changes at the Naval Shipyard site. First, as in any benefit transfer, the policy site does not match perfectly to the studies on which the meta-analysis was based, and the meta-model does not represent perfectly all characteristics of the Naval Shipyard location. Additionally, important characteristics of the Naval Shipyard area, such as its importance as a shipping location and harbor, are not included in the meta-model.

3.3.2.3 Salt Marsh Health

Salt marsh provides a number of ecological services with economic value, include flood control, storm buffering, water filtration, fish and wildlife nursery habitat, sediment stabilization, etc. Individuals may also value the knowledge that local salt marsh is healthy and will be preserved for future generations. We estimated the economic value of changes in the quality and quantity of salt marsh habitat at the Naval Shipyard site by (1) identifying measures of ecological effect and exposure from the Naval Shipyard risk assessment that are likely to be correlated with the economic services provided by salt marsh habitat, (2) using those measures to estimate the change in economic services provided by salt marsh in the study area, and (3) using a benefit transfer approach to evaluate the total value of the change in salt marsh services.

The Naval Shipyard risk assessment presents a variety of ecological measures that may be correlated with the ability of salt marsh to provide economic services. Table 3-11 lists economic services provided by salt marsh, and for each service, list ecological measures from the Naval Shipyard risk assessment that might be relevant. For each measure, the table summarizes the hypothesized relationship with relevant ecological services with economic value, and presents our estimate of the change in the quality or quantity of the services that can be inferred from the measure. Note that we selected services for inclusion in this table based on the services valued in Woodward and Wui (2001), a wetlands valuation study that is discussed below.

Table 3-11: Ecological Endpoint Measures Associated with Salt Marsh Economic Services

Ecological Service with Economic Value	Measure of Ecological Effect or Exposure	Relationship with Economic Service[a]	Data and Results from Risk Assessment[b]	Effects on Salt Marsh Ecosystem Health[c]	
				Change in Service	Explanation
Quantitative Evidence					
Acres of salt marsh	Salt marsh grass cover	Medium confidence.	Potential effect; cover of *S. alterniflora* was 15% less in two marsh sites than at reference sites. Other sites had similar or greater levels of cover (6-21, 6-55).	-15% to -5%	Range estimated based on observed change in cover.
Other Supporting Evidence					
Acres of salt marsh	Mollusk abundance	Low confidence; increased mollusk abundance could result in increased grazing of salt marsh (Silliman and Bertness, 2002).	Higher abundance observed in some marshes compared to reference sites. At marsh sites with higher abundance, mollusk abundance was 19% greater (6-21, 6-55).	Higher abundance may be indicative of potential effect.	
Ability to provide water quality control	Cover of vascular plants other than salt marsh grass	Low confidence; higher vascular plant cover may indicate lowered ability for water quality control.	Vascular plant cover was present in four out of nine marsh sites while present in only one out of seven reference sites. However, where present, vascular plant cover was 1% to 1.6%, compared to 0.2% at reference site (6-21, 6-55)	Change in vascular plant cover may be indicative of a potential effect on water quality control ability.	
	Contaminant concentration in bulk sediments	Higher levels of contaminant concentration may indicate that sediment is net source of contaminants instead of a net sink. However, high levels of sediment concentration may also be indicative of the continued ability of sediment and peat to capture contaminants.	Concentrations of several toxic metals were higher at the study sites than at the reference sites (IX.5-9)	Effect on salt marsh is ambiguous.	
Ability to provide on-site or off-site support for recreational and commercial fishing	Mollusk abundance	Mollusks serve as a prey species for recreational and commercial fish, including flounder. However, mollusks can also destroy salt marsh habitat.	Potential positive effect; higher abundance observed in some marshes compared to reference. At marsh sites with higher abundance, mollusk abundance was 19% greater (6-21, 6-55)	Presence of mollusks could be indicative of either an increase or decrease in ability to support fish populations.	
	Amphipod abundance	Amphipods serve as a prey species for recreational and commercial fish, including flounder.	Amphipod abundance in low marsh areas was four times below reference levels at all study sites. Abundance in middle marsh and high marsh was comparable with reference sites (6-57).	Some negative effect is likely.	

Ecological Service with Economic Value	Measure of Ecological Effect or Exposure	Relationship with Economic Service[a]	Data and Results from Risk Assessment[b]	Effects on Salt Marsh Ecosystem Health[c]	
				Change in Service	Explanation
Ability to provide bird habitat	Contaminant concentration in salt marsh grass leaf tissue	Contaminated salt marsh grass tissue may indicate risk to herbivorous birds, since it is a measure of the likely contamination levels of other plant food sources.	Contaminant concentrations were above the reference level at only one of the study sites.	The effect is likely to be small, if present at all.	
	Amphipod abundance	Amphipods serve as a prey species for birds, as well as forage for fish that are eaten by birds.	Amphipod abundance in low marsh areas was four times below reference levels at all study sites. Abundance in middle marsh and high marsh was comparable with reference sites (6-57).	Effect is possible, but difficult to quantify.	
	Mollusk abundance	Mollusks serve as prey for some avian species.	Potential effect; higher abundance observed in some marshes compared to reference. At marsh sites with higher abundance, mollusk abundance was 19% greater (6-21, 6-55)	Effect is likely to be small.	
General aesthetic quality	Salt marsh grass morphology	Many morphology characteristics, such as plant height and richness, may strongly affect the aesthetic quality of salt marsh areas.	No significant effects observed.	No effect indicated.	
	Number of animal taxa	The presence of a variety of animal taxa may be considered desirable.	Results mixed. Fewer taxa were identified at some study sites than at the reference sites, but in other study sites, more were found.	Results ambiguous.	
Ability to provide support for terrestrial and avian species with non-use values	Amphipod abundance	Amphipods serve as prey for terrestrial and avian species.	Amphipod abundance in low marsh areas was four times below reference levels at all study sites. Abundance in middle marsh and high marsh was comparable with reference sites (6-57).	Effect is likely, but difficult to quantify.	
	Mollusk abundance	Mollusks serve as prey for terrestrial and avian species. They may also compete with other species for food.	Potential effect; higher abundance observed in some marshes compared to reference. At marsh sites with higher abundance, mollusk abundance was 19% greater (6-21, 6-55)	Effect is possible, but difficult to quantify.	
	Contaminant concentration in salt marsh grass leaf tissue	Contaminants in leaf tissue can contribute to dietary exposure by birds (medium confidence).	HQ's for dietary exposure of Canada geese and black ducks through *Spartina* were well below one for each contaminant. However, combined Hazard Index was closer to one (0.8), indicating cumulative dietary exposure to multiple contaminants in *Spartina* may impact these birds (6-34)	Effect is likely to be small.	

Ecological Service with Economic Value	Measure of Ecological Effect or Exposure	Relationship with Economic Service[a]	Data and Results from Risk Assessment[b]	Effects on Salt Marsh Ecosystem Health[c]	
				Change in Service	Explanation
[a] Confidence levels reflect the strength of the association between ecological measure and salt marsh health indicated in the Naval Shipyard risk assessment, as well as in available literature on the relationship between the ecological measure and economic endpoint. [b] References in parentheses refer to section and page numbers in the Naval Shipyard risk assessment report. [c] Note that the percentage changes described in this column are best estimates of the entire change in water quality, based on each measure. Thus, these percentages changes are not cumulative across measures. The total change in water quality should be calculated using the average of these percentage changes.					
Source: U.S. Navy (2000).					

We monetized the estimated changes in the services provided by Naval Shipyard salt marsh using a benefit transfer methodology. To estimate total value per acre for salt marsh at the study site before and after contamination, we used the results of a meta-analysis by Woodward and Wui (2001). As a sensitivity analysis, we also estimated the value of the lost salt marsh acreage using the results of Mazzotta (1996), a valuation study conducted for the Peconic Estuary in New York.

Results based on Woodward and Wui (2001)

Woodward and Wui (2001) presents the results of a meta-analysis of a variety of wetlands valuation studies. The meta-model from this study estimates the total annual value of wetland, on a per acre basis, based on information about the services provided by the wetland and information about the methodology used to estimate the total value. Table 3-12 presents the meta-analysis variables and coefficients, as well as the values assigned to each variable. For detailed information about the variable definitions, refer to Woodward and Wui (2001). The following bullets briefly explain how we assigned values to each of the variables for our estimate of baseline value per acre:

- The intercept (*Intercept*) was set to 1.
- Study year (*Year*) was set to the mean for the meta-analysis dataset (note that *Year* = 0 corresponds to 1960).
- *Ln acres*, the natural log of the number of acres of wetland being evaluated, was set to 4.79 (the natural log of 120 acres).[4]
- Selected wetland characteristics dummy variables (*Coastal*, *Quality*, *Quantity*, *Rec. Fish*, *Com. Fish*, *Birdhunt*, *Birdwatch*, *Amenity*, and *Habitat*) were set equal to one to reflect a coastal wetland that provides water quality control, support for recreational and commercial fishing, support for bird hunting and watching, aesthetic enjoyment for nearby residents, and habitat for species with non-use values. The remaining wetland characteristics variables (*Flood* and *Storm*) were set to zero.
- Study methodology variables (*Publish*, *Data0*, *Theory0*, and *Metric0*) were assigned values that reflect a published study using reliable data, theory, and econometric

[4] Refer to Appendix C for documentation of our estimate of the total number of acres of salt marsh at the Naval Shipyard study site.

techniques. *PS* was set to zero to reflect a study that does not estimate producer surplus. *NFI* was set to one, reflecting a study that uses the net factor income method to estimate total value, since this is the preferred method of the authors of the meta-analysis. All other variables representing alternative valuation methodologies such as hedonic pricing (*HP*), replacement cost (*RC*), and travel cost (*TC*) were set to zero.

To calculate value per acre in the post-contamination (current) scenario, we used the information from Table 3-11 to modify the input value for the wetland area variable (*ln_acres*). We then calculated new values per acre.

Based on the variable assignments shown in the table, average value per acre for salt marsh at the study site is $704.17 in the baseline.[5] However, in the post contamination scenario, value per acre falls to $660.47 or $688.72, for the high and low acreage loss scenarios, respectively.

Table 3-12: Estimating Value per Acre for Salt Marsh

		Assigned Value		
			Post-Contamination	
Variable	**Coefficient**	**Baseline**	**Lower Bound**	**Upper Bound**
Intercept	7.872	1	1	1
Year	0.016	14.9	14.9	14.9
Ln acres	-0.286	4.79	4.62	4.74
Coastal	-0.117	1	1	1
Flood	0.678	0	0	0
Quality	0.737	1	1	1
Quantity	-0.452	0	0	0
Rec. Fish	0.582	1	1	1
Com. Fish	1.36	1	1	1
Birdhunt	-1.055	1	1	1
Birdwatch	1.804	1	1	1
Amenity	-4.303	1	1	1
Habitat	0.427	1	1	1
Storm	0.173	0	0	0
Publish	-0.154	1	1	1
Data0	0	0	0	0
Theory0	-1.045	0	0	0
Metric0	-3.186	0	0	0
PS	-3.14	0	0	0
HP	5.043	0	0	0
NFI	0.273	1	1	1
RC	2.232	0	0	0
TC	-0.341	0	0	0
Total Value per Acre per Year (2003$)		**$704.17**	**$660.47**	**$688.72**

[5] Note that values estimated from the regression equation have been converted from 1990$ to 2003$.

		Assigned Value
Source: Woodward and Wui (2001).		

The final step in our calculation of the economic value of changes in salt marsh ecosystem function at the Navy Shipyard site was to multiply the number of acres before and after contamination by our estimated value per acre before and after contamination. Table 3-13 shows the results of these calculations. Based on a total of 120 acres in the baseline and 102 acres to 114 acres after contamination, we estimated that the total annual value of the loss of salt marsh acreage could range from -$17,133 to -$5,986.

Table 3-13: Economic Value of Loss of Salt Marsh Acreage, Based on Woodward and Wui (2001)

Scenario	Annual Value per Acre (2003$)	Total Acreage	Total Annual Value of Existing Marsh (2003$)	Total Annual Value of Loss of Marsh (2003$)[a]
Baseline				
Average	$704.17	120	$84,500	n/a
Post-Contamination				
Lower bound	$660.47	102	$67,368	-$17,133
Upper bound	$688.72	114	$78,514	-$5,986
[a] Total annual value of loss of marsh is calculated by subtracting the total annual value of the baseline marsh from the total annual value of the post-contamination marsh.				

There are several significant limitations and uncertainties associated with using the Woodward and Wui (2001) meta-analysis model to estimate the value of salt marsh at the Naval Shipyard site. First, as in any benefit transfer, the policy site does not match perfectly to the studies on which the meta-analysis was based, and the meta-model does not represent perfectly all characteristics of the Naval Shipyard location. Additionally, important characteristics of the Naval Shipyard area, such as the number of local residents who have non-use values for salt marsh or who make use of salt marsh services, are not included in the meta-model. Furthermore, we have not attempted to adjust the regression equation to account for changes in services provided by salt marsh in the Naval Shipyards area. Finally, we note that the results of the Woodward and Wui (2001) regression model are not very robust. The model has a R^2 value of only 0.582, and only nine of the 23 variables in the model are significantly different from zero at a ten percent level of confidence. Thus, values predicted with this model will include a substantial degree of uncertainty.

Results based on Mazzotta (1996)

For comparison with the results from the analysis based on Woodward and Wui (2001), we also conducted a benefit transfer using the results of another valuation study: Mazzotta (1996). This study used an original contingent choice survey to estimate the relative preferences of residents and second homeowners living near the Peconic Estuary in New York state for preserving and restoring salt marsh habitat. The results indicate that non-user households in the Peconic study area are willing to pay approximately 5.5 cents per acre per year to preserve wetlands (in 2003$).

We applied the results from Mazzotta (1996) to the Naval Shipyards case study by assuming that households in the Naval Shipyards case study area have non-use values for wetland habitat that are similar to the values expressed by non-user households in the Peconic Estuary area. We then multiplied these per-acre per-household values by the number of acres of wetland lost in the Naval Shipyards study area, and by the number of households within 25 miles of the study area.[6] Table 3-14 shows the results of these calculations. Total non-use values to preserve six to 18 acres of wetlands are $0.33 to $0.99 per household, respectively. Based on an affected population of 85,624 households who live within 25 miles of the Naval Shipyards study area, the total non-use value of the lost wetlands is $28,256 to $84,768. Even though these values include only non-use values, they are higher than the total values calculated using the Woodward and Wui (2001) meta-analysis model, which include both use and non-use values.[7]

Table 3-14: Economic Value of Loss of Salt Marsh Acreage, Based on Mazzotta (1996)

Scenario	Change in Number of Acres	Annual Non-use Value per Acre per Household (2003$)	Annual Non-use Value per Household (2003$)	Total Number of Households	Total Annual Non-use Value of Loss of Marsh (2003$)
Lower bound	-6	$0.055	-$0.33	85,624	-$28,256
Upper bound	-18	$0.055	-$0.99	85,624	-$84,768

There are several important limitations and uncertainties associated with using the results from Mazzotta (1996) to estimate the value of salt marsh at the Naval Shipyard site. Many are similar to the limitations discussed above for Woodward and Wui (2001). As in any benefit transfer, the policy site (the Naval Shipyards site) does not match perfectly to the valuation study site (the Peconic estuary). However, the study does evaluate salt marsh in the northeastern United States in an area that is geographically and ecologically similar to the Naval Shipyards policy site. Although use of the study in an unadjusted single-site benefit transfer may introduce uncertainty into the resulting benefit estimates, given the similarity of the site characteristics, the benefit transfer exercise is likely to yield meaningful results.

3.3.3 Total Value

Table 3-15 summarizes the results of this valuation exercise. The table shows that the change in social welfare from contamination of the Naval Shipyard site has a lower bound of -$2.32 million and an upper bound of -$0.42 million. Since much of the supporting evidence that we evaluated indicated only a relatively low likelihood of environmental harm, total damages are likely to be closer to the small end of the range.

[6] Our estimate of the number of affected households within 25 miles of the Naval Shipyards study site is documented in Appendix B.

[7] Although Mazzotta (1996) provides WTP values for both users and non-users, we chose not to use this study to estimate total values for wetlands losses in the Naval Shipyards study area. Since only some households in the Naval Shipyards area are likely to be considered users of estuarine habitat in that area, and since Mazzotta's reported WTP values for users are only five percent higher than values for non-users (5.8 cents per acre per household compared to 5.5 cents per acre per household, respectively), we assumed that the use value of wetlands resources in the Naval Shipyards area is negligible compared to the non-use value of those resources.

The table shows non-use values for water quality are by far the largest affected category of economic value. For example, -$2.14 million of the -$2.32 lower bound change in social welfare is attributable to changes in water quality. This finding has important consequences for analysis of economic values associated with ecological changes. In particular, it shows that accurate estimation of all categories of value is not always necessary. If one category of economic value is known to be much larger than other categories of value, more analytical effort should be devoted to developing and refining value estimates for that category. In this case, refinement of the estimates of the value of changes in salt marsh function or recreational fishing would not have added much to the precision of the analysis, since these categories of value are less than eight percent of the total.

Table 3-15: Summary of Change in Social Welfare at Naval Shipyard Study Area

Type of Value	Change in Social Welfare (2003$)			
	Lower Bound	**Low Middle**	**High Middle**	**Upper Bound**
Use Values for Recreational Fishing	-$94,315	-$35,368	-$11,789	$0
Non-use Values for Water Quality[a]	-$2,143,517	-$1,003,855	-$876,734	-$410,593
Total Value of Salt Marsh Ecosystem Services	-$84,768	-$28,256	-$17,133	-$5,986
Total, All Evaluated Categories	**-$2,322,600**	**-$1,067,480**	**-$905,656**	**-$416,579**

[a] Multiple values for each scenario represent values based on alternative estimates of WTP per individual.

4. Challenges, Recommendations, and Future Research Needs

The results of the valuation exercise presented in the previous section highlight a number of challenges associated with valuation of ecological services. However, the exercise also demonstrates techniques for dealing with these problems, and suggests directions for future research activities.

4.1 Challenges Associated with Valuation of Ecological Changes

The primary problem that arose during our valuation exercise was an ecological modeling problem: how can available measures of ecological function be used to estimate changes in ecological services that have value to humans? Ecological endpoint measures used in risk assessments are often chosen because of their sensitivity to ecological disturbance or because they are easy or inexpensive to measure. Unfortunately, none of these criteria are likely to ensure that chosen measures have clear links with economically valuable ecological services. For example, in our analysis of impacts on recreational fishing, we were unable to use any of the available ecological measures (such as pollutant concentrations in sediments and in fish livers) to estimate changes in fish populations. The ecological measures do provide evidence that fish populations have been exposed to chemicals that have been shown to have reproductive effects. Furthermore, the weight of the evidence suggests a potential, though likely small, impact on reproductive success, which in turn could translate into effects on population size and subsequent recreational fishing success. However, it was very difficult for us to quantify this relationship. While some models exist to approximate the effect of changes in ecological indicators on populations of fish and other economically valuable ecological services, this case study, as with many ecological risk assessments, did not perform those modeling exercises, but instead relied on ecological indicators linked conceptually to these services. Rationally linking these indicators quantitatively with outcomes is one of the key challenges associated with valuation of ecological benefits.

A second major problem that we encountered was the difficulty of finding economic valuation studies relevant to ecological services of interest. Some kinds of economic benefits, such as use values associated with recreational fishing, are well documented in the academic literature. However, for some other types of benefits, such as the non-use benefits associated with eelgrass or avian community health, few high-quality valuation studies are available. Nonetheless, comprehensive economic analysis of ecological changes requires valuation research covering a variety of types of ecological systems and services.

4.2 Methodological Recommendations

The valuation exercise presented in Section 3.3 demonstrates that it is possible to provide economic information that is relevant to the evaluation of potential ecological changes, even when ecological data are limited. In addition, the experience with this exercise suggests several specific recommendations that may improve future analyses of ecological benefits.

First, evaluations of ecological benefits should acknowledge and attempt to quantify as many uncertainties as possible. Two aspects of uncertainty highlighted by our analytic methodology are particularly important. First of all, we estimated of a *range* of possible ecological changes and economic values. Second, we carefully and explicitly documented all assumptions used to generate the results. These two actions help reviewers to understand the level of uncertainty involved and the limitations of the resulting estimates.

Second, future ecological benefit evaluations should consider using a preliminary economic screening analysis. As shown in the valuation exercise, some categories of ecological benefits may be significantly larger than other categories. In particular, non-use values for improved water quality were an order of magnitude larger than all other economic values combined. These results suggest that future studies of ecological benefits might find it useful to conduct initial screening analyses using worst-case damage assumptions for each category of benefits, and utilizing, to the degree possible, results from previous analyses of similar resources. If the results of such an analysis indicate that certain categories of ecological benefits are significantly smaller than other categories, little added value might be gained from expending effort and resources to refine estimates of value for these categories. Instead, resources can be devoted to analysis of larger, more important categories of value.

A final point from valuation exercise is that small modifications to the design of future ecological risk assessments could significantly improve the ability of analysts to link the risk assessment results to economic values. In particular, selection of ecological measures with clear and well-defined relationships with economically valuable ecosystem services, and the use of these measures together with well-established population and community-level modeling techniques (such as matrix models), would greatly facilitate future economic analysis.

4.3 Research Needs

The limitations and challenges discussed in this and previous sections highlight a number of gaps in current knowledge about ecological risk and the economic value of ecological services. Future research in a number of subjects is needed to address these gaps.

First of all, many past research efforts have focused on the recreational, commercial, and non-use value of changes in fish populations or changes in water quality. However, less research has been done to quantify the economic value of other types of ecological resources, such as the recreational value of lobster and shellfish fishing and bird hunting, or the non-use value associated with different types of common animals, plants, and habitats. Additional research on these topics would broaden the spectrum of ecological endpoints than can be linked to economic values.

Another general subject area that is poorly understood is whether individuals place greater value on preserving overall ecosystem health or preventing specific ecological disturbances. Future research on this topic would help researchers to understand the relative importance of different aspects of ecological benefits that are valued by individuals.

Finally, as discussed above, quantifying the relationship between easily measurable ecological endpoints and economically relevant ecological endpoints is a key challenge for future research.

Quantitative research is needed to establish empirical relationships between convenient ecological measures and economically important ecological services.

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APPENDIX A: Estimating Recreational Catch

We estimated the change in recreational catch at the Naval Shipyard study site by first calculating total baseline recreational catch at the study site, based on data from the National Marine Fisheries Service's Marine Recreational Fisheries Statistics Survey (MRFSS). According the MRFSS, 2.03 million fish are caught each year in Maine, and 0.75 million fish are caught each year in New Hampshire (the Naval Shipyard site is located on the border between the two states). We then estimated the fraction of these fish that are caught at the study site by assuming that total catch in a marine area (excluding big game, which are unlikely to be affected by nearshore contamination) is proportional to the number of miles of shoreline in that area. For example, to calculate the fraction of the total catch in Maine that is caught in the Naval Shipyard study area, we divided 2 miles (the approximate length of the Naval Shipyard shoreline that is located in Maine) by 228 miles (the length of the entire Maine coast). We then multiplied the result, 0.9%, by the total catch in Maine to calculate catch at the Maine portion of the study site. We followed the same procedure for New Hampshire. Tables A-1 and A-2 document the results of these calculations. Overall, we estimated that 17,783 fish in Maine and 41,789 fish in New Hampshire are caught near the Naval Shipyard study area, for a total of 59,572 fish.

Table A-1: Percentage of Statewide Catch Occurring at Naval Shipyard Study Site

State	Miles of Coastline in State	Miles of Coastline at Naval Shipyard Site	Percentage of Catch Likely To Be Caught at Naval Shipyard Site
Maine	228	2	0.9%
New Hampshire	18	1	5.6%
Source: TravelNotes.org (2005); Wikipedia (2005).			

Table A-2: Total Catch at Naval Shipyard Study Site, By Species

MRFSS Species Group	Meta-Analysis Species Group	Total Catch in State	% of Catch Near Study Site	Catch Near Study Site
Maine				
Eels	Other Saltwater	376	0.9%	3
Wrasses	Other Saltwater	16,975	0.9%	149
Bluefish	Small Game	55,776	0.9%	489
Herrings	Other Saltwater	7,664	0.9%	67
Sculpins	Other Saltwater	2,637	0.9%	23
Catfishes	Other Saltwater	8,425	0.9%	74
Searobins	Other Saltwater	179	0.9%	2
Other fishes	Other Saltwater	58,766	0.9%	515
Cods and hakes	Other Saltwater	226,695	0.9%	1,989
Temperate Basses	Small Game	778,031	0.9%	6,825
Tunas and mackerels	Big Game	848,777	0.0%	0
Cartilaginous fishes	Other Saltwater	23,015	0.9%	202

MRFSS Species Group	Meta-Analysis Species Group	Total Catch in State	% of Catch Near Study Site	Catch Near Study Site
Total, All Species		**2,027,316**		**10,338**
New Hampshire				
Eels	Other Saltwater	86	5.6%	5
Wrasses	Other Saltwater	2,868	5.6%	159
Bluefish	Small Game	31,983	5.6%	1,777
Herrings	Other Saltwater	2,861	5.6%	159
Sculpins	Other Saltwater	10,908	5.6%	606
Flounders	Flatfish	12,060	5.6%	670
Searobins	Other Saltwater	196	5.6%	11
Sea basses	Other Saltwater	346	5.6%	19
Other fishes	Other Saltwater	4,117	5.6%	229
Cods and hakes	Other Saltwater	274,797	5.6%	15,267
Temperate basses	Small Game	206,380	5.6%	11,466
Tunas and mackerels	Big Game	97,714	5.0%	0
Cartilaginous fishes	Other Saltwater	107,880	5.6%	5,993
Total, All Species		**752,196**		**36,360**
Source: NMFS (2003).				

APPENDIX B: Estimating Number of Affected Households

To calculate the number of households within a 25-mile radius of the Naval Shipyard study site, we used a map of census tracts from the U.S. Census Bureau in combination with the U.S. EPA's Reach File of waterways (U.S. Census Bureau, 2000; U.S. EPA, 1996). We loaded these data into a geographical information system (GIS), which we used to plot the center of each census tract and to calculate the 25-mile radius surrounding the study site. We calculated the total number of households near the study area by summing the number of households in each census tract whose center fell within the 25-mile radius surrounding the study site. The result of this analysis showed that 85,624 households are located with 25 miles of the Naval Shipyard site.

The decision to include all households within 25 miles of the study site was somewhat arbitrary. In all likelihood, many households located further than 25 miles from Naval Shipyard may hold non-use values for changes in water quality at the site. Empirical studies in economic literature suggest that individuals are likely to hold non-use values for both local and regional resources (Pate and Loomis, 1997; Schulze et al., 1995). Thus, this assumption may provide a conservative estimate of the total affected population near the Naval Shipyard study site.

APPENDIX C: Estimating Salt Marsh Acreage

Our estimate of the total number of acres of salt marsh potentially affected by contamination at the Naval Shipyard site is based on our estimate of the number of miles of coastline at the site (see Appendix A for documentation of this estimate). We made the very conservative assumptions that the entire three miles of coastline is salt marsh, and that the salt marsh cover is 110 yards (1/16 of a mile) wide along the coast. Based on these two assumptions, we calculate that there are 0.1875 square miles, or 120 acres, of salt marsh in the Naval Shipyard area.