Ecology and Economics of Compensatory Restoration

by

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Table of Contents

1.0 Introduction

Natural ecosystems provide a vast suite of ecosystem services that sustain the functioning of the ecosystems themselves and enrich human experience, enterprise, and endeavor (Daily *et al.,* 1997). For example, as part of a co-evolutionary partnership honey bees pollinate many terrestrial shrubs and trees, facilitating their propagation, while also serving to pollinate the flowers of orchard trees and field crops essential to agricultural production. Ecosystem services come for free and easily go unrecognized and unappreciated. As a consequence, environmental insults to public waters and lands and the ecosystems they support were long accepted without requiring compensation. Oil spills, chemical releases, habitat destruction during land development, and many other human-induced perturbations to natural ecosystems represented an externalization of the cost of doing business, with the public implicitly losing when degraded services were not replaced by the parties responsible.

In the United States, numerous federal policies and legislative acts have recognized the wisdom of preserving critical ecosystem services provided by public trust resources and have established requirements for the compensation of environmental injuries. A "no net loss" policy for wetlands under the federal Clean Water Act requires that construction projects avoid, then minimize where unavoidable, and finally mitigate any remaining damage to wetlands. The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and the Oil Pollution Act (OPA) establish procedures for federal and state governments and American Indian tribes to assess injuries to natural resources caused by oil or chemical releases and to seek compensatory restoration for losses to the public trust system. The authority to seek compensation for natural resource damages (NRD) under these statutes has led to a burgeoning field of study that is the source of most techniques described in this document. Other United States laws with a restoration component include the Coastal Zone Management Act, the Endangered Species Act, the Marine Mammal Protection Act, and many others (Allen *et al.*, 2005). Requirements for restoration and public compensation for resource injuries are now being adopted in the European Union under the Environmental Liability Directive of 2004 (EU, 2004).

Here we describe a selection of concepts, models, and techniques that have been developed over the past three decades in the United States for determining appropriate restoration to compensate for lost ecosystem services. Compensatory restoration involves a series of steps, beginning with identification of the ecosystem services that are affected, then progressing to selection of proxies (metrics) representing the most important ecosystem services, quantification of injury using those proxies, and development of equivalency models that demonstrate how alternative restoration projects offset ecosystem losses. These techniques collectively determine the appropriate scale of compensatory restoration actions and are often known as "restoration scaling" methods. We explain the logical basis in ecology and economics that underlies restoration scaling methods. We examine numerous techniques that have been developed to address impacts to a wide range of habitats, organisms, and human activities. We describe how compensation can be achieved through substitution of lost ecological services by different but related services using conversions based on biological function. We explore the role of public value in determining equivalency between injured and restored services, and in accounting for time delays between the loss and replacement of ecosystem services through the use of economic discounting. We also provide reference to our compiled data bases of (1) case studies on restoration scaling and (2) numerous primary sources in the ecology and economics literature that have contributed to the current status of the theory and practice of compensatory restoration scaling.

This document is organized based on the division of ecosystem services into two main categories: ecological services and human-use services. Ecological services support the interrelated functions of natural communities and provide the basis for human-use services, supporting many human enterprises and values. Human-use services provide an essential input to the conduct of human activities such as outdoor recreation and commercial navigation. We begin in Section 1.0 with an introduction to the methods and concepts of restoration scaling common to both types of ecosystem services. Section 2.0 describes methods used to determine appropriate compensation for ecological losses, such as habitat equivalency analysis, resource equivalency analysis, and survey-based resource valuation methods. We go beyond a description of the modeling concepts to provide a synthesis of techniques specifically developed for particular habitats and resources. Section 3.0 describes methods used to analyze human-use losses from environmental impacts. In a field that has been explored in numerous published sources, we focus specifically on concepts and methods essential to practitioners and researchers in the NRD arena and likewise applicable to compensatory mitigation for planned and permitted ecosystem degradation. Finally, section 4.0 describes the rationale and applied techniques used in economic discounting of ecological and human-use losses.

1.1 Compensatory restoration

Compensatory restoration refers to any human intervention in the environment that increases the net benefits provided by natural resources and is used to offset a loss of resource benefits. Resource benefits are often called "services", though it should be clear in this context that in addition to changes in the resource characteristics themselves we also refer to changes in the ability to obtain benefits from resources, such as through the purchase of wilderness lands for public access. Service losses requiring compensation may be caused by such human interventions as toxic contamination that results in the death of birds, an oil spill that disrupts recreation at a public beach, or commercial development that displaces wetland habitat. Service gains may be achieved through projects such as protection of land from development, the placement and seeding of oyster shells in an estuary to establish oyster reef habitat, or the removal of a dam to enhance salmon spawning runs.

Figure 1.1 shows graphically the role of compensatory restoration in offsetting lost resource services. When an incident occurs that causes resource injury, services decline to below their baseline level. "Baseline" refers to the level of services the resource would have provided through time had the injury not occurred. The baseline level of services varies over time and thus is not necessarily the same as the level of services prior to injury because of temporal variations in natural environmental forcing factors and other human interventions impacting resource services. Efforts may be undertaken to minimize the further spread of or directly remediate the resource injury, and these efforts are often termed "primary" restoration. Primary restoration accelerates recovery to baseline, reduces the total quantity of services lost over time, and reduces the required quantity of compensatory restoration. Area A under the baseline services curve in Figure 1.1 shows the total loss in services that accrues prior to recovery, sometimes described as "interim" loss. Note that this interim loss would have been greater in the absence of the primary restoration done to contain the damage.

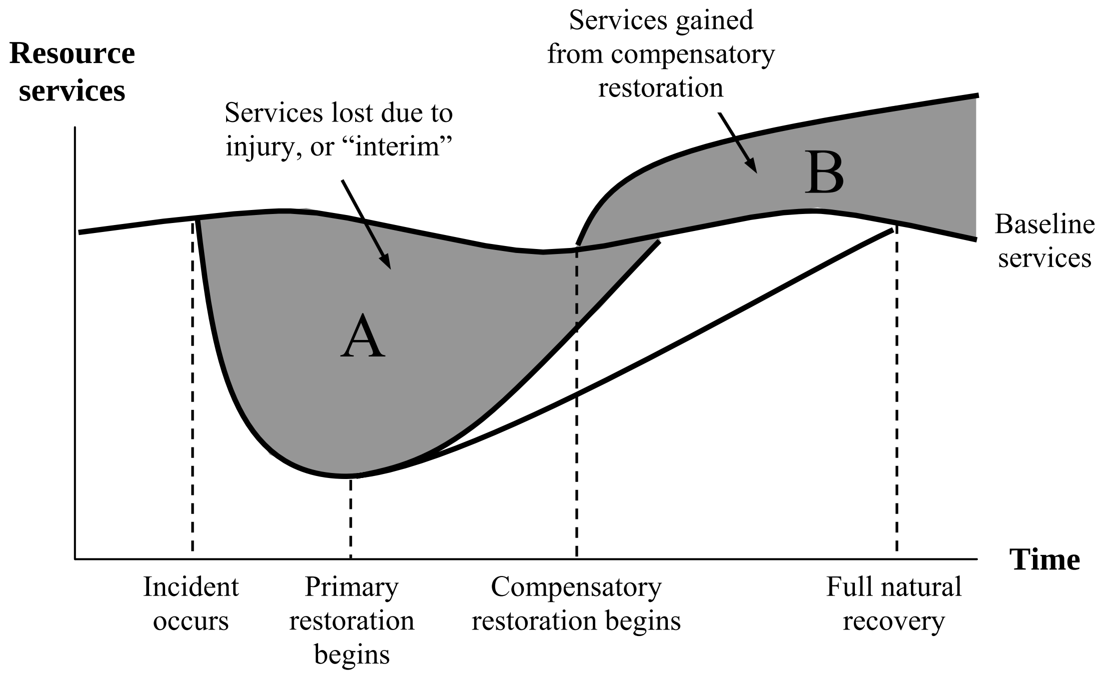

Figure 1.1. Using compensatory restoration to offset lost resource services

Compensatory restoration projects are selected, and compensatory restoration is implemented, at some time following the initial incident. Compensatory restoration increases resource services to above their baseline level and thereby holds potential for offsetting the interim losses. Compensatory restoration is distinguished from primary restoration in that it enhances services different from those injured, with the difference being either the type of services restored or the location where services are restored. Baseline service flows with respect to restoration are thus distinct from the injury baseline, but are illustrated in the graph using a single line for simplicity. The difference between restored and baseline services for compensatory projects traces out area B, representing the total gains from compensatory restoration over time.

Restoration scaling methods determine appropriate compensation by establishing equivalence between services lost in area A and services gained in area B. Restoration scaling is based on the formula

$$\sum_{t=0}^{T} \frac{1}{(1+r)^t} L_t = \frac{V_g}{V_l} \sum_{t=0}^{T} \frac{1}{(1+r)^t} G_t \,. \qquad (1.1)$$

L_t is a measure of losses due to injury in period *t*, such as a decline in the breeding success of birds in a contaminated floodplain. G_t is a measure of gains from restoration in period *t*, such as an increase in breeding due to preservation of alternative breeding habitat. Losses and gains are quantified over a sequence of time periods to capture the changing impacts of injury and restoration over time, and to account for the influence of timing of service flows on the present value of resource services. From the base period 0 through the final period *T*, service losses and gains are multiplied by a discount factor $1/(1+r)^t$ to convert period-*t* services to their equivalent value in period 0. The term *r* is the discount rate, which quantifies the declining importance of future resource changes when evaluated in the present, as described in section 4.0. Discounted services are summed across periods for a comparison of total losses and gains.

The measures used for L_t and G_t are specific to the particular characteristics of injury and restoration, and the term V_g/V_l controls for differences in value between units used to measure losses and units used to measure gains. Specifically, it is the ratio of the value of one unit of service gains V_g to the value of one unit of service losses V_l. Sometimes the ratio V_g/V_l is explicit, as when a loss of upland habitat is replaced by an expansion of wetland habitat and the latter is determined to have a higher value by a factor of 2.5 to one (Bailey Trustee Council, 2003). In other cases the ratio of value is implicit, as when mortality of turtle hatchlings is offset by an increase in restored turtle hatchlings and the one-to-one ratio of value drops out of the equation (Byrd *et al.,* 2002). Sometimes losses and gains are explicitly evaluated in monetary terms, for example when recreational services are evaluated using economic models of recreation choice (Lavaca Bay Trustee Council, 1998). In this case value conversions occur not from one type of service to another but from each type of service into a common currency unit such as dollars. The monetary value of losses and gains is more conveniently expressed by rearranging equation (1.1) to include the term $V_l\,L_t$ on the left side and $V_g\,G_t$ on the right side.

1.2 Modeling approaches

Methods for quantifying service losses induced by injury and service gains emerging from restoration require the development of an appropriate metric. Like most modeling techniques, the selection of a metric involves a tradeoff between simplicity and realism. A metric must be simple enough to be observed with reasonable precision and at reasonable cost. However, selection of a simple metric necessarily limits consideration of multiple complex factors relevant to the realistic evaluation of ecological services. In the NRD arena, four basic modeling approaches have been defined to address compensatory restoration scaling, and each is associated with a particular type of injury and restoration metric. The approaches are known as resource-to-resource, service-to-service, value-to-value, and value-to-cost.

The resource-to-resource modeling approach quantifies injury and restoration based on impacts to the biologically based entities of primary interest, such as habitats or organisms. If the primary impacts of an oil spill are smothering of mangroves and loss of the habitat they provide or the death of sea turtles, injury could be measured as the number of acres of mangrove lost or the number of turtles killed. Preservation of mangrove habitat and the elimination of threats to turtles such as discarded fishing gear could also be measured using units based on the resulting increases in acres of habitat and number of organisms, respectively. L_t and G_t in equation (1.1) would thus be direct measures of the ecological changes of interest, namely, changes in the quantity of valued resources. Methods that use the resource-to-resource approach are typically called habitat equivalency analysis (HEA) or resource equivalency analysis (REA). Although the HEA and REA terminology is not consistently applied in the literature, this manuscript will use the term HEA for methods specifically applied to habitat changes and REA for methods applied to organisms.

The service-to-service modeling approach overcomes limitations in the ability to measure injury and restoration as changes in resource quantity. Habitats in particular may be degraded but not lost entirely, and changes in a particular habitat service may be a useful measure of the severity of degradation. For example, river-bottom habitat provides a food source for fish by supporting populations of benthic organisms, but contamination of sediments limits the ability of river-bottom habitat to provide this ecological service. If studies measure a 15-percent mortality rate in benthic organisms attributable to contamination, this quantifiable ecological change could be used as a metric for the decline in all services provided by the contaminated river-bottom habitat. Other examples of service metrics include indices of species diversity, vegetative percent cover, or measures of organism function such as reproductive success for sentinel species present in a particular habitat.

Value-to-value scaling differs from resource-to-resource and service-to-service scaling by explicitly accounting for value to the public in measuring resource changes. Specifically, resource changes can be valued in monetary terms based on the public's willingness to pay to enhance environmental services or prevent ecological injury. Willingness to pay is measured using surveys conducted with a sample of the general public in which respondents choose among alternative resource management options. These "stated-preference" surveys may also elicit the public's value for tradeoffs between resource injury and restoration without explicitly measuring value in monetary terms. Outdoor recreation is an important service provided by natural resources, and models of recreation behavior such as travel-cost analysis can also be used to evaluate injury and restoration. Value-to-value methods are especially important in addressing ecological impacts for which biological metrics are difficult to identify, for example, sub-lethal effects to organisms that do not readily lead to a measurable change in the total quantity of a resource.

Value-to-cost scaling is a modification of value-to-value scaling in which the cost of a project is used as a proxy for its value. The value-to-cost approach is typically used when a monetary estimate of losses is readily obtained but a monetary valuation of compensatory improvements is difficult. For example, a decline in fishing trips following an oil spill may be measured using aerial counts of anglers accessing the spill area, and multiplying the decline in trips by a per-trip value obtained from the economics literature provides an estimate of the monetary loss to fishing. Dam removal may be selected as a compensatory project because it will increase fish populations and benefit recreational anglers, but a precise quantification of the increase in value would require a costly study of the resulting increase in catch rates and its value to recreational anglers. As long as benefits from the dam removal project at least equal the cost of the project, then using project cost as a proxy for value ensures that compensation is achieved. For projects of modest size, it is common practice in NRD and throughout the public arena to rely on the judgment of resource managers to determine whether a project's benefit exceeds its cost. Value-to-cost scaling is discussed in greater detail in the context of human-use losses, where it is most frequently applied.

1.3 Ecology and economics

Compensating for resource losses requires quantitative methods to measure changes in ecosystem services. Because lost and compensatory services differ in their location, type, or timing, the measurement of losses and gains requires judgments of value that cannot be captured solely by ecological parameters. Techniques to determine appropriate compensation require an approach that combines ecological factors with economic theory and methods.

The role of ecology and economics in restoration scaling is illustrated by the distinction between primary restoration and compensatory restoration. Primary restoration involves returning resources to their baseline condition. The success of primary restoration can be determined using solely biological parameters, and it is possible for metrics based on economic value to differ from science-based metrics. For example, even if people placed no value on the destruction of vegetation in a seagrass bed, it would still be possible to identify the existence of an injury and to monitor the return to baseline conditions using biological metrics such as above-ground and below-ground plant biomass.

There is no such distinction between value-based metrics and science-based metrics in the context of compensatory restoration. Once baseline conditions are restored, there is no purely ecological measure of the appropriate level of additional services that would compensate for past losses. This is because the notion of compensatory restoration is not meaningful in a purely ecological context, and is instead based on economic principles. If people placed no value on seagrass beds, there would be no need for compensatory restoration following resource injury. If people place a value on certain services provided by seagrass beds but not on others, then the appropriate metric for restoration scaling will address the valued services. The study of ecology is important to understanding connections between the various components of a habitat and the resource services they provide. However, the conceptual validity of a metric for scaling compensatory restoration depends on its ability to capture changes in public value associated with alternative resource conditions.

The economic motivation for compensatory restoration arises from principles of equity and efficiency. Compensation is a matter of equity because pollution represents a potentially unjust redistribution of value from the public, in the form of lost environmental quality, to the polluter, in the form of financial gain. Compensation is also a matter of economic efficiency, as embodied in the "polluter pays" principle. The polluter-pays principle states that a company should bear the full environmental costs associated with any polluting activities. The reason is that economic welfare is maximized when companies have an incentive to refrain from taking any action whose total costs, including environmental costs, exceed total benefits. Only when polluters pay compensation for environmental harm will they fully account for both costs and benefits when making economic decisions.

Compensation for interim losses is consistent with the principles of equity and efficiency if it makes members of the public as well off as they would have been in the absence of the pollution incident. Expressed another way, compensation should make the public indifferent between a situation in which an incident occurs and compensation is provided, and the alternative situation in which the incident does not occur. These principles are readily applied to the analysis of human-use services, but they are equally important to the analysis of compensation for lost ecological services. Some of the implications of an economic understanding of ecological scaling are addressed in Sections 2.6 and 2.7.

The combined role of ecological expertise and public value significantly complicates the practice of compensatory restoration scaling. Some methods discussed in this document emphasize the importance of ecological expertise by relying solely on the judgment of scientists to determine appropriate compensation. Most assessments using HEA or REA fall into this category. The difficulty with this approach is that scientists, especially those who have chosen to work in the field of natural resource damage assessment, are a self-selected group whose preferences may differ significantly from those of the general public. This means that the values that they place on resource injury and available restoration alternatives could differ from the values held by members of the public to whom compensation must be provided. The study of public preference is replete with examples of the folly of ignoring self-selection (e.g., "Dewey Defeats Truman") and practitioners who study public choice go to great lengths to ensure a valid representation of the affected population.

Unfortunately methods to ensure that public preferences are adequately represented may fail to address important ecological issues. For example, stated-preference techniques use surveys of the general public to determine the appropriate amount and type of restoration to offset resource injury. Survey materials present respondents with information about important ecological factors, such as the impacts of contamination on plants and wildlife and natural variation in baseline conditions. The extent to which respondents become sufficiently familiar with the relevant ecological issues is the subject of debate among researchers. The motivation to continue improving survey-based methods to evaluate ecological services arises from the importance of public choice as the foundation for the compensation of resource losses.

2.0 Ecological Losses

Restoration scaling for ecological losses often involves the use of biological metrics to quantify losses and gains in resource services. Habitat Equivalency Analysis (HEA) relies on metrics such as vegetative cover or sediment toxicity to evaluate a change in habitat services. Resource Equivalency Analysis (REA) involves an organism rather than a habitat, and almost always involves animal mortality or sub-lethal reproductive effects resulting in lost offspring. The metrics in REA models usually involve numbers of organisms or total biomass of a population of organisms. Both HEA and REA rely on a common analytical structure that leads many practitioners to use the terms interchangeably, but for clarity we will maintain the distinction between HEA and REA throughout this document. Another important scaling approach involves the use of public surveys to evaluate resource services. These "stated preference" methods may evaluate losses in purely monetary terms, but more recent applications directly estimate tradeoffs between resource injuries and restoration actions. HEA and REA models represent the most widely used tools for restoration scaling, but stated-preference methods have been examined far more thoroughly in the literature due to their widespread use in the broader environmental policy arena.

2.1 Ecological equivalency models: HEA and REA

Compensation for interim losses was traditionally viewed as a claim for monetary damages (Yang *et al.,* 1984; DOI, 1986). With the recovery of injured resources addressed through primary restoration, the motivation for addressing interim losses was primarily economic, involving compensation, incentives to deter polluting activities, and fairness. The need to address interim losses is particularly clear when the best course of action at an affected site is to allow for natural recovery, so that without interim losses liability for resource impacts would be zero (Brans, 2001). A practical solution was to calculate the value of interim losses and spend the recovered funds on resource enhancements. Difficulties with this approach included the high cost of studies to value resource injury and controversies surrounding the available economic methods, such as contingent valuation (Arrow *et al*., 1993).

Recognizing the high cost of resource valuation studies, Unsworth and Bishop (1994) proposed HEA as an alternative approach to determining compensation for interim losses. Their published article formalized methods that practitioners in the field had been developing over several years. The HEA model that they developed involved a set of economic assumptions that allowed monetary values to be replaced by units of habitat area. The emphasis shifted from a damages-based claim to a compensation-based claim, with losses in habitat area replaced by gains in habitat area. The amount of a claim for compensation was thus based on the cost of restoring the required habitat area, and the exercise of converting habitat losses to a monetary value was no longer necessary. The HEA method was upheld in court in *United States v. Fisher* (1997) and *United States v. Great Lakes Dredge and Dock* (2001).

Subsequent studies refined the HEA model and replaced simple measures of habitat area with more precise biological metrics of habitat service and function (Fonseca *et al.,* 2000; Strange *et* *al.,* 2002; Cacela *et al.,* 2005). Studies such as Penn and Tomasi (2002) and Zafonte and Hampton (2005) helped expand the method from a habitat model (HEA) to a resource model (REA) that addressed impacts such as mortality to birds or fish. Like habitat area in the original HEA model, the expanded set of metrics measured biological changes that were assumed to be directly proportional to changes in public value, so that the exercise of monetizing value could be avoided. In 1996, regulations for the Oil Pollution Act expressed an explicit preference for HEA/REA models over monetary valuation (NOAA, 1996). However, more recent regulations issued under the Superfund law (CERCLA) refrained from affirmatively encouraging HEA or REA over methods based on monetary valuation (DOI, 2008).

The basic elements of the HEA/REA model can be described with reference to the scaling equation (1.1) in Section 1.1. Losses due to injury L and gains from restoration G are calculated using biological metrics. Changes in metrics relative to a baseline level represent changes in the value of resource or habitat services. In many cases G and L are measured in the same units, so that $V_g = V_l$ and lost services are replaced by services of the same type. This is sometimes called "in-kind" restoration scaling. Adjustments may be made to G and L to ensure that $V_g = V_l$. For example, when the baseline level of services at a restoration site differs from baseline at the injury site, restoration services are measured as a percent of baseline at the injury site to ensure consistency in units of measure. Likewise, REA methods often adjust for differences in age-class structure of injured and restored populations. Given these adjustments to G and L to equate V_g and V_l, models of this type are also referred to as in-kind scaling. Whenever V_g does not equal V_l, the ratio V_g/V_l explicitly appears in the scaling equation and the approach is called "out-of-kind" scaling. The ratio V_g/V_l represents a conversion between different habitats or resources. Conversions may be calculated using the relative productivity of two or more habitats, estimates of trophic efficiency relating a restored food source to increases in a food-limited population, or other methods discussed in Section 2.5.

Units used in metric-based scaling incorporate the basic elements of equation (1.1). All units are discounted, services are frequently quantified by habitat area (often acres), and the T periods are frequently expressed as years. This leads to the use of discounted service-acre-years (DSAYs) as the most common scaling unit. One DSAY of salt marsh habitat is equal to the value of services provided by one acre of salt marsh in year 0. The value of one acre of salt marsh provided at some later date has a year-0 equivalent that is less than one DSAY due to the effect of discounting. Often habitat services are measured not as the presence or absence of habitat, but as changes in the level of habitat function. If a restoration project increases services in one acre of salt marsh from a level that is 50-percent of full function to a level that is 80-percent of full function in year t, then the year-t value of restored service flows is 0.3 service-acre-years. The resulting increase in DSAYs would be $0.3/(1+r)^t$. When restoration scaling is applied to organisms rather than habitat, services are frequently quantified by numbers or biomass of organisms. In this case the appropriate units might be discounted bird-years, discounted kilogram-years, or some other analogous unit.

The use of biological metrics, conversion factors, and discounted service flows is explored in the many applications that follow. For other useful introductions to HEA/REA methods, refer to NOAA (1995), Hampton and Zafonte (2002), Penn and Tomasi (2002), and Allen *et al.* (2005).

Automated software to perform HEA/REA calculations is available at the web page for Nova Southeastern University's National Coral Reef Institute (Kohler and Dodge, 2006).

2.1.1 HEA/REA calculations: hypothetical example

This section illustrates the HEA/REA model using a simple example that combines elements from selected NRD cases. This complete description of a restoration scaling exercise from start to finish may provide a helpful context for the discussion of case-specific scaling techniques that follows. This stylized example includes illustrations of the assumptions and judgments often required in scaling models, and it provides references where relevant calculations are presented in greater detail. The steps are described below and summarized in Tables 2.1 to 2.4.

Scenario: A barge releases 100,000 gallons of oil several miles offshore. Oil washes ashore in an area characterized by a mixture of sand flats and salt marsh. Post-spill reconnaissance indicates that patchy oiling occurred throughout both types of habitat. Oiled, dead birds are also observed in some areas. A fate and transport model that accounts for water currents, winds and physical properties of the released oil (e.g., French McCay *et al*., 2004) indicates a low probability that significant oil would have mixed into the water column or sunk to the sea bottom before dissipating to below harmful levels.

A loss of ecological services is assessed for marsh, sand flats and birds. A 75 percent initial service loss is estimated for oiled areas of marsh based on the observed decline in vegetative cover in these areas compared to oil-free reference sites (e.g., Michel *et al.,* 2002). The total area of oiled marsh is ten hectares. Ten hectares of sand flats are also assigned a 75 percent initial loss. Scientists survey the shoreline and recover 100 birds that appear to have died from exposure to oil. The count of dead birds is multiplied by five based on the likelihood that many injured birds would wash out to sea or escape detection in visual surveys (Sperduto *et al.,* 2003).

Table 2.1. Quantifying Habitat Losses

			Marsh – 10 ha			Sand Flats – 10 ha		
Period	Time Since Spill (years)	Discount Factor	Year-End Service Loss per ha	Mid-Year Service Loss per ha	Total Discounted Service Losses (DSHaYs)	Year-End Service Loss per ha	Mid-Year Service Loss per ha	Total Discounted Service Losses (DSHaYs)
Time 0	0		75%			75%		
Year 1	0.5	0.99	60%	67.5%	6.7	0%	37.5%	3.7
Year 2	1.5	0.96	45%	52.5%	5.0	-	-	-
Year 3	2.5	0.93	30%	37.5%	3.5	-	-	-
Year 4	3.5	0.90	15%	22.5%	2.0	-	-	-
Year 5	4.5	0.88	0%	7.5%	0.7	-	-	-
Total					17.8			3.7

Note: The discount factor is $(1+r)^t$, where r is the 3-percent annual discount rate and t is the time in years elapsed since the spill. Discounted services losses are calculated as midyear losses multiplied by the discount factor and the area of injured habitat. DSHaYs are discounted service-hectare-years.

Future losses and the path to recovery are estimated. Based on monitoring of marsh recovery following previous spills, the oiled marsh is expected to return to baseline condition in five years (e.g., Michel *et al*., 2002; Penn and Tomasi, 2002). Service losses are assumed to decline linearly from 75 percent at the start of the first year to zero at the end of the fifth year. Sand flats are expected to return to the baseline level of services in one year. The loss in ecological services captured in bird mortality is expected to end after a single year because of the ability of the bird population to rebound in that time. The likelihood of second-generation losses for birds due to forgone reproduction is determined to be small enough to ignore (NOAA *et al*., 2002; Zafonte and Hampton, 2005).

Future losses are discounted into present-value terms. Table 2.1 shows the pattern of habitat losses through time and the discounting of losses to time zero when the spill occurred. Annual losses are discounted using a three-percent discount rate. For example, the rate of service loss for marsh is 22.5 percent at the midpoint of the fourth year following the spill. Multiplying by ten hectares of oiled marsh, service losses in the fourth year would be expressed as 2.25 service-hectare-years. Discounting this figure requires multiplying it by the discount factor $1/(1.03)^{3.5}$, or 0.90. The exponent 3.5 refers to the time in years between the date of the spill and the midpoint of the fourth year. The discounted value of fourth-year losses is therefore 2.0 discounted service-hectare-years (DSHaYs). The same midyear adjustments and discounting procedures are applied to sand flat habitat. Discounted service losses are summed across years to calculate a total loss of 17.8 DSHaYs for marsh and 3.7 DSHaYs for sand flats.

Losses due to bird mortality are summarized in Table 2.2. Losses are confined to a single year and are constant up to the next annual breeding cycle. For this one-year loss the timing of service flows and the effects of discounting are insignificant and are ignored. The observed mortality of 100 birds is extrapolated to 500 birds, as noted above. Multiplying by a weight of two kilograms per bird results in 1,000 kg-years of birds lost.

Table 2.2. Quantifying Losses Due to Animal Mortality

Observed dead birds	100
Estimated ratio of observed bird mortality to total bird mortality	1:5
Estimated dead birds not observed	400
Total estimated bird mortality	500
Average weight of a bird (kg)	2
Biomass measure of bird mortality (kg)	1,000
Time to recovery of bird population (years)	1
Bird mortality losses (kg-years)	1,000

Discounted losses for all injured resources are expressed as a loss to marsh using resource conversions. As shown in Table 2.3, losses to sand flat habitat and birds can both be converted to equivalent marsh losses using appropriate conversion factors. The ecological value of a hectare of sand flat is estimated to be one-fifth the value of a hectare of marsh. This conversion is based on the relative productivity of the two types of habitat, the function of each habitat as refuge for animals, and other factors (Westchester Trustee Council, 2000). The loss of 3.7

DSHaYs of sand flats is therefore equivalent to a loss of 0.74 DSHaYs of marsh. The conversion rate between birds and marsh is based on the annual food production of marsh habitat compared to a bird's annual food intake (French McCay *et al*., 2002). As outlined in the table, the annual secondary production of a typical marsh in the area of the spill is 5,147 kg/ha/year. Secondary production refers to small sediment-dwelling invertebrates that directly or indirectly consume plant matter from the marsh. Marsh production is multiplied by 0.4 percent to account for trophic conversion losses in biomass as invertebrates are eaten by fish and fish are eaten by birds. We assume for this example that the birds that suffered mortality in this incident were secondary predators like herons and egrets that eat small fishes. This results in a figure of 20.6 bird-kg-years generated by one hectare of marsh in a year. The loss of 1,000 bird-kg-years is therefore divided by 20.6 to obtain 48.6 DSHaYs of marsh-equivalent service losses. Total marsh-equivalent service losses associated with bird mortality, sand flat injury and marsh injury are 0.74 + 48.6 + 17.8 = 67.2 DSHaYs.

Table 2.3. Resource Conversions

Injury to sand flats	
Service losses (DSHaYs)	3.7
Estimated ratio of ecological services per ha, sand flats to marsh	1:5
Marsh-equivalent losses (DSHaYs)	0.74
Bird mortality	
Service losses (kg-years)	1,000
Annual salt marsh secondary production (kg/ha/year)	5,147
Ecological efficiency between secondary marsh production and birds	0.4%
Total bird production per ha of marsh (kg/ha)	20.6
Marsh-equivalent losses (DSHaYs)	48.6
Marsh service losses (DSHaYs)	17.8
Total marsh-equivalent losses (DSHaYs)	67.2

The discounted gain in ecological services from a selected marsh restoration project is estimated. To offset spill-related losses, new wetlands are to be created from an abandoned golf course bordering a nearby estuary. The golf course is built on filled marsh and excavation of

Table 2.4. Quantifying Restoration Benefits

			Marsh Creation – 4.85 ha		
Period	Time Since Spill (years)	Discount Factor	Year-End Service Gain per ha	Mid-Year Service Gain per ha	Total Discounted Service Gains (DSHaYs)
Time 0	0		0%		
Year 1	0.5	0.99	0%	-	-
Year 2	1.5	0.96	0%	-	-
Year 3	2.5	0.93	0%	-	-
Year 4	3.5	0.90	5%	3%	0.1
Year 5	4.5	0.88	11%	8%	0.3

-	-	-	-	-	-
Year 17	16.5	0.61	75%	72%	2.1
Year 18	17.5	0.60	80%	77%	2.2
-	-	-	-	-	-
Year 50	49.5	0.23	80%	80%	0.9
Total					67.2

earth from the site would restore natural elevations and tidal flows and allow the reestablishment of marsh habitat. Prior to restoration, ecological services at the site are estimated to be 10 percent of those of a fully functioning marsh. Restoration would occur in the third year after the spill. Following restoration, service flows are expected to increase from 10 percent to 90 percent over a 15-year period, resulting in an 80-percent net service increase (Strange *et al*., 2002). The 80-percent service increase is then expected to remain constant through the 50th year following the spill. Due to uncertainties associated with effects of sea-level rise and other factors, potential gains accruing from the project beyond 50 years are not included in the estimate of restoration benefits. A three-percent discount rate is applied to future service flows. The project covers an area of 4.85 ha. The timing of service flows and the discounting calculations are shown in Table 2.4.

Total discounted losses and gains are determined to be equivalent. As shown in Tables 2.3 and 2.4, the total benefits of restoring 4.85 ha of marsh are equivalent to the sum of spill-related losses associated with marsh, sand flats and birds. Ecological losses of 67.2 DSHaYs are fully compensated by the marsh restoration project. The scale of the marsh project could be adjusted to compensate for alternative levels of injury.

2.2 Quantifying losses in habitat equivalency analysis (HEA)

This section presents a variety of metric-based models for determining service losses resulting from habitat injury. Methods are organized according to habitat type, including coastal marsh, sedimentary benthic habitat, water column, oyster reef, mangroves, seagrass, coral reef, kelp forest, and other habitats. HEA modeling relates habitat service losses to gains from restoration. Methods for establishing the relationship between metrics used in injury quantification and techniques for determining appropriate compensation are described in Section 2.4, which investigates methods for quantifying restoration benefits, and Section 2.5, which describes conversions between different types of resources.

2.2.1 Coastal marsh

Coastal marshes lie at the interface between the terrestrial and aquatic realms, exposing them to terrestrial run-off high in nutrients, sediments, and contaminants. Consequently, this habitat is highly productive per unit area, yet threatened by land development pressure and vulnerable to deposition of floating pollutants like oil. Coastal marshes of the continental U.S. differ regionally as a function of variations in coastal tidal hydrology, geomorphology, human encroachment, and biotic province. Much of this variation is related to regional differences in geologic legacy and disturbance regime. The most extensive marshes occur along the Gulf of Mexico coast, especially in Louisiana and Florida, and along the Middle and South Atlantic coast; the least extensive are along the North Atlantic, and marsh acreage per unit of ocean shoreline is lowest on the Pacific coast, particularly in Central to Southern California (NOAA, 1990). Whereas the highest percentage loss (90%) of historic coastal wetlands has occurred in California (NOAA, 1990), the greatest losses in marsh area have occurred in Louisiana and Florida (extrapolated from total wetland loss statistics in Dahl, 1990).

Marsh plant communities are low in species diversity and the typically few dominant species tend to be notably stress tolerant (Stout, 1984; Bertness and Ellison,1987; Mitsch and Gosselink, 2000; Sullivan and Currin, 2000). North America as a whole is exceptional for the large number of terrestrial vertebrate taxa that are endemic or largely restricted to tidal marshes (Greenberg *et al.,* 2006). Because of often dramatic differences in elevation and subsequent flooding frequency and duration, tidal marshes exhibit substantial spatial heterogeneity in ecosystem services. Tidal marshes are valued, protected, and restored in recognition of the comprehensive suite of their ecosystem services: (1) high productivity and habitat provision supporting the food web leading to fish and wildlife (Teal, 1962; Weisberg and Lotrich, 1982; Boesch and Turner, 1984; Peterson and Turner, 1994; Minello *et al.*, 2003); (2) buffers against storm wave damage (Mitsch and Gosselink, 2000); (3) shoreline stabilization; (4) flood water storage (Mitsch and Gosselink, 2000); (5) water quality maintenance; (6) biodiversity preservation (Keer and Zedler, 2002; Callaway *et al.*, 2003); (7) carbon storage and biogeochemical cycling; and (8) socio-economic benefits. While marsh protection and restoration have a long history in the United States, incentive programs to convert farm fields into marsh habitat are spreading in European Union countries also (Crépin, 2005).

In this section, we first review the Chalk Point case (NOAA *et al*., 2002), which represents an advanced and well-documented methodology for quantifying injury in forms appropriate for scaling compensatory restoration of marsh habitat. We then review an approach to treating different marsh habitat types and associated non-marsh habitat for the purpose of quantifying restoration, using the Bailey Trustee Council (2003) case. Finally we consider a study of what metrics best represent the suite of ecosystem services provided by marsh habitat, looking ahead to where injury assessment may progress in the future.

2.2.1.1 Quantifying marsh injury using metrics suitable for scaling in-kind habitat restoration

The Chalk Point oil spill case illustrates a current methodology for quantifying salt marsh injury. Injuries to about 76 acres of brackish marsh, as well as other connected estuarine habitats, resulted from a spill of No. 2 and No. 6 fuel oil from a ruptured underground pipe in the marsh along Swanson Creek on the shore of Chesapeake Bay (NOAA *et al*., 2002). Field surveys along with aerial photography of the marsh demonstrated that 23.4 acres were heavily oiled, 12.0 acres moderately oiled and 40.5 acres lightly oiled. Surveys conducted immediately after the April 2000 spill, then in July and September, and then again in the subsequent July 2001 provided data on oiling appearance on the soil surface, vegetation, and subsurface of the soil, the degree of oiling (coverage and thickness), vegetative metrics of marsh grasses (stem height and density, percent cover), sediment chemistry (marsh surface soils evaluated for levels of total petroleum hydrocarbons and PAHs), and abundance and composition of benthic macroinfauna in 1-m^2 quadrats established in both oiled and adjacent reference areas. These data were used by reference to previous oil impacts to estimate the degree of injury and time trajectory for full recovery, required to compute injury in units of lost discounted acre-years (DSAYs) of marsh ecosystem services. To capture two categories of ecosystem services provided by marsh habitat, both marsh vegetation and soil condition were used as dual metrics to determine degree of injury caused by the spill. Above-ground marsh vegetation reflects a wide range of ecological functions related to primary production, trophic support, habitat structure, fish and shellfish production, as well as recreational and aesthetic value. Marsh soils are important for habitat for invertebrates, long-term support of marsh plants, and biogeochemical cycling.

The marshes were divided into strata reflecting oiling degree and vegetative habitat. Oiling was defined as light, moderate, or heavy. Light oiling referred to less than 10% cover by oil in the initial survey and an oil thickness of < 0.01 cm. Moderate oiling had a coverage of > 10% (average of 60%) and oil thickness > 0.01 cm on marshes outside of the Swanson Creek source area. For the light oiling category, all wetland types were combined into a single category. For moderate oiling, the two *Spartina* species were combined because of very similar responses. Heavily oiled wetlands (those with > 10 % oil cover, > 0.01 cm oil thickness, and location within Swanson Creek) were divided into 6 strata, with shoreline and interior areas for each of the three dominant vegetation types (*Typha sp., Spartina alterniflora,* and *S. cynosuroides)*. The percent service loss for vegetative services and soil services plus the recovery trajectory was then estimated for each marsh vegetation stratum by professional judgment. Lightly oiled wetlands (all species combined) were judged to have suffered only a 10% loss in both soil and vegetation services, with complete recovery in 6 months. Moderately oiled *Spartina* marshes were judged to have experienced an initial 50 percent loss in function, with an expected recovery of 1 year for vegetation and 3 years for soils.

Heavily oiled strata were assigned higher levels of service losses for both vegetation and soils, with soil recovery rate differences between edge and interior locations, based on the reasoning that greater tidal exchange would induce faster dissipation of oil in the edge locations (NOAA *et al.*, 2002). Specifically, heavily oiled *Typha* was assigned a 100% initial service loss to vegetation services in both interior and edge zones, with recovery assumed complete in 1 year. Heavily oiled *Typha* was assigned soil losses of 75% in the edge, recovering to a 40% loss after 3 years and fully recovering after 10 years. Heavily oiled *Typha* in the interior marsh was assigned soil losses of 50% initially, recovering to a 20% loss after 5 years with complete recovery after 10 years. For heavily oiled *Spartina alterniflora*, initial vegetation service loss was assumed to be 100%, with 50% recovery after 1 year and full recovery after 5 years in both interior and edge positions. Soil services were assumed to suffer initial losses of 75% in both interior and edge marshes, with interior recovering to a level of only 25% loss after 5 years and full recovery after 10 years, and edge recovering to a level of 20% loss after 3 years and complete recovery after 5 years. For heavily oiled *Spartina cynosuroides*, initial vegetation service loss was assumed to be 100%, with 50% recovery after 1 year and full recovery after 10 years in both interior and edge positions. Soil services were assumed to suffer initial losses of 75% in both interior and edge marshes, with interior recovering to a level of 50% loss after 5 years and full recovery by 20 years, and edge recovering to a level of 40% loss after 3 years and full recovery after 10 years. Vegetation- and soil-related ecosystem service losses were independently estimated from this injury and recovery scheme in units of lost discounted service acre-years (DSAYs). Assuming that vegetation and soil services of wetlands are equally important, the joint ecosystem service loss was computed by a simple average of the vegetation service loss and the soil service loss in DSAYs.

Michel *et al*. (2008) describes a follow-up assessment of how the vegetation and soil recovery had progressed 7 years after oiling of the Chalk Point marshes. This follow-up monitoring took place only in the heavily oiled interior portions of the *Spartina alterniflora* and *Spartina cynosuroides* habitats. Results can be compared against analogous surveys made just after the spill in April 2000, and then in July 2000 and July 2001.

Three metrics were recorded during the 2007 resampling (Michel *et al*., 2008): (1) persistence and weathering status of PAHs in soils at 0-10 and 10-20 cm depths; (2) vegetation condition (below-ground biomass, live stem density, and stem height); and (3) surficial soil toxicity in amphipod (*Ampelisca abdita*) bioassays. The results of soil PAH analyses demonstrated that PAH weathering of soils in both species of marsh grass had not advanced at all in the 7 years since initial sampling after the spill. Furthermore, sediment toxicity testing revealed that half the soil samples were still toxic after 7 years. These results suggest that the recovery trajectories assigned during injury assessment were conservative, since they presumed that soil services would recover from 25% in 2000 to 75% in *S. alterniflora* and 50% in *S. synosuroides* after 5 years and to 100% after 10 and 20 years, respectively. Vegetation sampling in 2007 revealed that *S. alterniflora* had 37% lower stem densities and 15% lower stem heights and *S. cynosuroides* had 20-35% less below-ground biomass, as compared to predictions of complete vegetation recovery in *S. alterniflora* in 5 years and in *S. cynosuroides* in 10 years (NOAA *et al.*, 2002).

Another telling follow-up on long-term recovery dynamics of oiled salt marshes comes from the resampling of Cape Cod marshes over 40 years after the spill from the barge Florida. After 40 years, PAHs were detected at 10-20 cm depths in marsh soils in only moderately weathered condition. Furthermore, densities of fiddler crabs (*Uca* sp.) remained lower on oiled marshes than in controls and crabs displayed behavioral differences attributable to possible exposure to toxins. Specifically, burrowing depths of the crabs were only half as deep on the oiled marsh and crabs on the oiled marsh exhibited sluggish responses to threats relative to control crabs (Culbertson *et al.,* 2007). The soils of the oiled marsh were highly organic and, similar to those of the interior marsh at Chalk Point, probably largely impervious to oxygen, limiting microbial weathering processes. The weathering limitations are enhanced by the inability of fiddler crabs to dig normally deep burrows because this process of bioturbation can be valuable in oxygenating deeper sediments and inducing weathering of buried oil.

Evidence from follow-up monitoring at the Chalk Point site and similar evidence collected at the oiled Cape Cod marshes suggests that the recovery curves developed for the Chalk Point assessment may have underestimated the length of time required for the recovery of oiled marsh. As predicted by Teal and Howarth (1984), heavy oiling of fine-grained salt marsh interiors probably suppresses recovery of at least soil but also biota ecosystem services for as many as four decades or more.

2.2.1.2 Normalizing marsh sub-habitat types to estuarine marsh habitat

The Bailey case (Bailey Trustee Council, 2003) involved injury of several sub-habitats within a marsh system. The Bailey Waste Superfund Site is a former waste disposal facility in Orange

County, Texas. Contaminants [polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), and heavy metals] were found at high concentrations in the sediments and soils of seven different sub-habitat types. The geographic scope of injury of each sub-habitat was determined by analytical chemistry. The trustees also decided that the level of ecosystem service loss was 100% in each of these sub-habitats of the marsh system and that these losses are permanent. To facilitate the injury analysis and provide a metric of injury that could readily lead to compensatory restoration, the trustees decided to convert service acre-year losses (SAYs) of each sub-habitat to functionally equivalent SAYs of the brackish marsh sub-habitat because this one was the most extensively injured and it provided the highest level of services. The relative per-acre values of each of the seven injured sub-habitat types were determined by a "multiple attribute decomposition" process, in which a group of six wetland scientists with knowledge of the local ecosystem (three representing the Responsible Parties and three the natural resource trustees), rated each of the seven habitats from 0 to 10, based upon the perceived value of the joint ecosystem services they provide (e.g., primary productivity, habitat value, nutrient export, etc.). The means of these scores were used to normalize each sub-habitat SAY to an estuarine marsh equivalent, which allowed calculation of the total injury in units of brackish marsh acres. Applying this technique, the 3.26 acres of high-marsh habitat that suffered 100% loss, was scaled to have equal ecosystem services equivalent to 1.98 acres of healthy brackish estuarine marsh. Table 2.5 illustrates the method and exhibits the score values used to normalize impacts to all sub-habitat types to brackish estuarine marsh losses (Bailey Trustee Council, 2003).

Table 2.5 Method and rank-score values to normalize habitat impacts to brackish estuarine marsh, Orange County, Texas (Bailey Trustee Council, 2003)

Habitat type	Score 1	Score 2	Score 3	Score 4	Score 5	Score 6	Average score	Normalized Average
Brackish tidal marsh	10.0	9.3	10.0	10.0	10.0	9.7	9.833	1.000
High marsh	5.0	6.5	6.0	5.0	7.0	6.3	5.967	0.607
Freshwater marsh	9.0	7.3	7.6	8.0	7.0	7.7	7.767	0.790
Ponds	6.0	4.5	6.3	6.0	5.0	5.2	5.500	0.559
Ditch	5.0	3.5	4.6	3.0	5.0	4.3	4.233	0.431
Upland	2.0	5.3	4.0	4.0	6.0	2.7	4.000	0.407
Road	0.3	2.0	0.6	0.0	1.0	1.0	0.817	0.083

This injury assessment methodology resembles the habitat conversion approach discussed later in detail in that it provides a metric by which one (sub)-habitat is converted into another in contemplation of restoration. We present this approach separately because there are some fundamental differences. Here, all habitats that are converted are arguably sub-habitats of a marsh complex not habitats typically recognized as separate. In addition, the "multiple attribute decomposition" method depends on best professional judgment by a small group of experts. This renders the metric subjective. In contrast, habitat conversion methods rely on a quantitative metric, namely production (usually secondary production). Some may consider the application of this quantitative metric to be more defensible and rigorous; however, it uses but one of the many important ecosystem services on which to scale acre-years of services of one type of habitat against another. This service used, biotic production, is one of acknowledged high value, but it may not be reflective of the other important services.

2.2.1.3 Alternative marsh metrics

Most injury assessments of tidal marshes following oil spills include only: (1) documenting the areas of marsh covered by heavy, moderate, and light oiling; (2) measuring stem density and height, perhaps also areal cover, of each dominant vascular plant species within each oiling intensity category; (3) sampling sediments for chemical (petroleum hydrocarbons and PAHs) analyses and depth of contamination, followed by sediment toxicity assays if sediment contamination is high and likely to persist; and (4) collecting and counting any dead animals. Other common types of marsh injury involve chronic contaminants by persistent organic pollutants in Superfund sites, where the contaminants may not cause plant injury but lead to (1) analytic chemistry analyses to compute concentrations of toxicants to compare against known biological effects levels and (2) trigger sediment toxicity testing and toxicological analysis of growth, reproduction, and mortality impacts at higher trophic levels. Salt marshes provide many ecosystem services (MEA, 2005), so reducing metrics to vegetative production of dominant vascular plants and/or sediment injury omits many valued processes. An important set of questions arises. Are structural measures of above-ground vegetation of vascular plants the most suitable metric for assessing level of ecosystem services of salt marshes? And should other metrics be added to make this assessment more complete and indicative of either injury level or duration of injury, the two factors needed to estimate service losses?

Peterson *et al*. (2008a) reviewed several alternative or additional metrics that could provide further quantitative insight into other important ecosystem service levels in marshes. These alternative metrics included: (1) routine stratification of marsh habitat into edge and interior (as done in the Chalk Point 2002 injury assessment (NOAA *et al.*, 2002) with separate sampling in each stratum; (2) microphytobenthos abundance; (3) cotton-strip decomposition bioassays and other biogeochemical indicators; and (4) sum of production across consumer trophic levels; and (5) below-ground biomass of vascular plants. Of these, designing marsh sampling to cover edge and interior strata as a routine practice would provide much more resolution of injuries and ensure more confident compensation because the edge is typically much more productive than the marsh interior. In addition, although sampling for below-ground biomass is necessarily destructive and damaging, the addition of this information may provide more insight into duration of injury and future productivity. However, more basic research will be needed to develop confidence in this metric and what it implies. Peterson *et al*. (2008a) concluded that the present metric of above-ground structural plant density is the best single current indicator of marsh ecosystem services because it correlates with many of the ecosystem services, including primary production, structural habitat provision for fish and wildlife, protection of the shoreline from waves, interception of sediments, nutrients, and pathogens in stormwater flows, aesthetics, carbon storage, and other services. No alternative metric relates to so many processes of value in wetlands.

2.2.1.4 Metric-free impact assessment

In small oil spills, where the expense of site-specific injury assessments would be disproportionately high relative to the expected injury, injury to ecosystem services may often be estimated on the basis of observed oiling of the marsh with no quantitative injury assessment. For example, on the basis of previous research, a lightly oiled marsh may be assumed to have a

10 percent initial service loss followed by a six-month recovery. Penn and Tomasi (2002) describe the use of oiling categories as a scaling tool in the context of the Lake Barre oil spill assessment. The appropriate relationship between oiling categories and service losses has been examined in detail in several oil spill assessments, including the Chalk Point (NOAA *et al.*, 2002) case, although uncertainty persists. For the Lake Barre case, Penn and Tomasi (2002) describe four recovery trajectories for salt marsh in Louisiana based on the severity of marsh oiling. The four designations were developed in a collaborative effort between public officials and representatives of Texaco, the company responsible for the spill. The experts relied on previous spill-response experience combined with field observations during the first year of recovery. Information was collected periodically regarding the persistence of oiling on plants and soils as well as the condition of plants and the presence of invertebrate species in oiled and reference areas. Other resource services such as feeding habitat for birds and spawning habitat for fish were not specifically examined, but were assumed to decline and recover in proportion to the observed variables. The estimated level of injury and recovery trajectories are presented in Table 2.6.

Table 2.6. Oiling categories and injury to saltwater marsh, Lake Barre

Category of injury	Initial service loss	Time to full recovery
Light oiling	10%	4 months
Heavy oiling, low	40%	2 years
Heavy oiling, medium	75%	2 years
Heavy oiling, high	100%	20 years

In areas of light oiling ecological services were assumed to decline by 10% initially, and full recovery was assumed after four months. As in all categories presented, the functional form of the predicted recovery path was linear. Heavily oiled areas were divided in to three levels of severity. Areas with 40 percent and 75 percent initial service losses were predicted to recover in 2 years based on extent of recovery in the first year following the spill. For areas of 100 percent service loss where all above-ground vegetation was killed, the time required for full recovery was difficult to predict. Because the extent of this severely affected area was limited, the parties agreed to a 20-year recovery horizon that was viewed as a conservative upper-bound estimate. Recent monitoring of the Chalk Point recovery of heavily oiled marsh 7 years after the spill (Michel *et al*., 2008) raises questions about how truly conservative this assumption of a 20-year recovery is.

2.2.2 Benthos or sedimentary benthic habitat

Soft sediments can be found in several different types of bottom habitats, including salt marsh, mangrove, and seagrass, as well as in several types of sedimentary bottom that lack biogenic structural vegetation. It is this suite of habitats that lack emergent vascular vegetation that define this habitat category. The sedimentary benthic habitat, so defined, actually encompasses a range of systems of varying biological composition, productivity, and value for fish and wildlife. These benthic habitats change with depth, ranging from intertidal flats to shallow subtidal sediments with sufficient incident light reaching the bottom to support microalgal production on down to deeper subtidal bottom lacking primary production. The benthic habitats also vary with sediment character, with sand and mud representing the two most common alternatives.

However, some benthic sedimentary habitats contain mixes of sediment types, including shell, cobbles, or other larger rocks. The shallow sand and mud flats have relatively high habitat value, both for primary production of microphytobenthos but also for infaunal marine invertebrates that support high levels of foraging by shorebirds, ducks, and demersal crabs and fishes (Peterson and Peterson, 1979; Peterson; 1992). Finally, the function and value of these sedimentary benthic habitats also depend on the landscape setting of proximity to other biogenic structured habitats, where fish, crustaceans, and birds can find structural refuges, and which produce detritus, which benefits herbivorous invertebrates on these unvegetated flats (Seitz *et al.*, 2006). As a rule, the shallower the sedimentary habitats, the more productive they are and the more they serve uses of fish and wildlife (Peterson *et al.*, 2009). This suite of sedimentary habitats supports production by microphytobenthos, more in the intertidal zone than in the shallow subtidal, and this production has great value for herbivorous benthic invertebrates because the single-celled algae are eaten directly without passing through a microbial intermediary (Kneib, 2003). That direct trophic pathway is more energetically efficient than detritivorous trophic pathways leading from vascular plants to detritivores (Peterson and Peterson, 1979). The intertidal and shallow subtidal sedimentary habitat is a very significant feeding grounds for shorebirds, waterbirds, fishes, and crustaceans (Peterson and Peterson, 1979).

The sedimentary benthic habitat has traditionally been considered much less productive and less important than salt marsh, seagrass beds, and other obviously vegetated habitats. Much recent research has dispelled this notion by showing that the benthic microphytobenthos is highly productive (Sullivan and Curran, 2000) and the productivity of intertidal flats is not dramatically lower than that of *Spartina* marshes (Peterson *et al.*, 2009). Nevertheless, the aggregate ecosystem services provided per acre of unvegetated sedimentary benthic habitat are unlikely to be valued as high as other biogenic habitats. Furthermore, these unvegetated sediments are usually in large supply in estuarine and marine ecosystems, rarely representing a limiting habitat for important species or ecosystem processes that provide key services. For these two reasons, compensatory restoration and mitigation will rarely target creation of new unvegetated sedimentary habitat and most injury to this type of habitat will be converted by HEA habitat trade-off ratios to equivalent ecosystem services from one of the biogenically structured habitats instead (Peterson *et al.*, 2009). However, sea level continues to rise and bulkheading remains a widespread practice of preventing property loss from erosion of estuarine shorelines, so intertidal flats habitats will gradually disappear (Peterson *et al.*, 2008b), creating a future incentive for creation of estuarine intertidal sedimentary habitats in compensatory restoration.

Intertidal and shallow subtidal sedimentary benthic habitat can become contaminated by specific environmental incidents like oil spills, but the large majority of cases of sedimentary habitat contamination involve long-term chronic contamination from chemical releases, discharges, leaks, or seepages. This includes many Superfund sites and analogous areas of less serious contamination. Such sites are typically characterized by multiple types of chemical contaminants and long histories of contamination, posing challenges to injury assessment. Whereas many species of fish, birds, crustaceans and other wildlife like sea otters and sea turtles make use of sedimentary benthic habitats for foraging and other activities, probably the most efficient and encompassing method of quantifying injury to sedimentary benthic habitats is by the HEA approach of estimating the percent service loss associated with various concentrations of contaminants. Thus a fractional reduction in total ecological services is assigned when concentration of a contaminant exceeds a particular threshold level associated with a known negative effect on a sedimentary benthic invertebrate or fish or wildlife species that uses that habitat. While this approach ignores the human use services and non-consumptive services to humans provided by sedimentary benthic habitats, some of these tend to be more difficult to quantify and can be assumed to follow restoration of ecosystem function (Wolotira, 2002).

2.2.2.1 Chronic chemical contamination of sedimentary benthic habitats

The Commencement Bay Trustee Council (2002) NRD record including especially its Appendix D (Wolotira, 2002) provides an example of the basic methodology of quantifying injury to sedimentary benthic habitat from chronic contamination. The HEA approach involves a means of determining the DSAYs (Discounted Service Acre-Years) associated with observed levels of sediment contamination, acres contaminated, relative value of each benthic sedimentary habitat affected, and duration of injury. This approach is formally summarized in Cacela *et al.* (2005).

This approach relies upon determining the relationship between levels of contamination, usually from multiple contaminants, and the percentage loss of ecological services. Although natural habitats provide both ecological and human ecosystem services, this approach focuses on quantifying the ecological service losses and assumes that the human consumptive use and non-use services follow accordingly (Wolotira, 2002). Substances of concern (SOCs) are identified and then, separately for each one of them, the relationship between sediment concentration and loss of ecological services is determined from all available data bases on benthic community change, regulatory standards, and toxicological response thresholds. Any negative biological response reflects a service loss, ranging from biochemical responses of organisms, behavior, growth, reproduction, and mortality, to population and community changes (Wolotira, 2002). Whereas negative responses of any of these variables represents injury, the degree of service loss varies with the seriousness of the response variable that shows injury, the intensity of the response, and the numbers of species (determined by toxicity tests) showing a detectable response (Commencement Bay Trustee Council, 2002). By comparing SOC concentrations in sediments to the full suite of biological response information available, Apparent Effects Thresholds (AETs) can be identified and tabulated by contaminant. Such tables specify the percentage service loss associated with ranges of concentration for each individual contaminant. Toxicological response data for sediments are available, normalized either to dry weight of sediments or to organic content. The Commencement Bay Trustee Council (2002) argued compellingly that the appropriate way to quantify contaminant concentration for NRD chemical contamination assessments is by dry weight of sediments because several types of contamination involve simultaneous enhancement of organic content of sediments. Thus normalizing concentrations per unit organic content fails to recognize the full extent of toxic contamination that occurs in conjunction with an enhancement of organic content (Wolotira, 2002).

The Commencement Bay methods of assigning AETs to two specific SOCs, HCB (Hexachhlorobenzene) and PAHs (Polycyclic Aromatic Hydrocarbons), provide generic insight into this approach to determine injury to sedimentary benthic habitats (Wolotira, 2002). For HCB, the only available toxicity data involve benthic invertebrates, whereas for the better studied PAHs data are available for higher trophic levels, specifically fish. Consequently, percent service losses caused by HCB were capped at 20% on the basis of lack of evidence that the injury extended beyond the benthic invertebrate secondary producer trophic level. In contrast, PAH injuries rose to a maximum of 80% because information from English sole demonstrated that injury rises to higher trophic levels (Wolotira, 2002). Other SOCs possess similar variation in the extent of information available on impacts to the biota of sedimentary habitats. For example, particularly extensive information on PCBs and congeners of DDT has been tabulated by McDonald (1994). For this Hylebos Waterway contamination in Commencement Bay, the State of Washington had an extensive data base on sediment contamination and toxic effects, such that substantial amounts of evidence were available for setting AETs. In some other states, relevant information may need to be provided across states but within similar biogeographic and ecological regions. Some decisions made for this Commencement Bay injury assessment need to be reconsidered on a case-by-case basis for future applications. Specifically, constraining the magnitude of injury to 20% because of lack of available toxicological or empirical field evidence of injury to higher trophic levels must also stand the test of whether there is a reasonable basis of concluding that linkages do not extend higher. Lack of evidence may establish a bias in the form of an explicit placement of burden of proof on the defenders of public trust resources. Furthermore, the English sole does indeed represent a demersal fish associated with sedimentary habitats and one that is not very mobile. This facilitates matching contaminant concentration in sediments to fish responses. It also helps justify combining the fish injury into the sediment habitat injury. However, this method would not be appropriate if a mobile fish or other higher-trophic level consumer were injured by contact and uptake of sediment contaminants if even brief and limited exposure had serious negative impacts. In such cases, the sediment contamination could affect a much larger population of the consumer than the average numbers associated with the contaminated area. Thus, service loss could in theory, and perhaps often in practice, exceed 100% when measured as service loss to habitat rather than organisms.

The first AET threshold to establish for any SOC is the no-effect level, below which the chemical causes no loss of ecological services. The State of Washington maintains a data base containing 7 different response variables relating to the sediment-dwelling invertebrates: benthic invertebrate community composition, plus 6 bioassay data bases – echinoderm bioassay, Microtox TH, amphipod bioassay, *Neanthes* polychaete bioasaay, a bivalve bioassay, and an oyster bioassay (Wolotira, 2002). Because the bivalve bioassay includes oysters, the bivalve test results were used only if oyster results were not available. Thus for HCB, the results of six response variables were used to construct a series of thresholds, each of which represented passage into a range of successively greater percentage loss of ecological services. A level of 22 ppb was determined to be the zero effects threshold: the benthic community abundance and species diversity was reduced in all sediments in the Washington data base with HCB concentrations greater than 22 ppb. Above that concentration, a 5% service loss was assigned. At levels (70 ppb) when half of the assay data variables demonstrated exceedence of their no effect level, a 10% service loss was assigned: for exceedence of three quarters of the assays (130 ppb), a 15% loss and, for exceedence in all six tests (230 ppb), a 20% loss. No data on fish or other higher-trophic level consumer were available in the Washington data base on HCB toxicity, so 20% became the highest degree of injury attributed to HCB alone (Wolotira, 2002). For compounds with only invertebrate toxicology data but for which only 3, 2, or 1 type of assay results were available, percentage injuries were assigned using fewer thresholds: 5, 10, 20; 5, 20; and 5%, respectively.

For PAHs, toxicity test information for English sole was also available to complement the benthic invertebrate bioassay data bases for use in determining how to scale service losses of the sedimentary benthic habitat to this contaminant. Unfortunately, the invertebrate AET data base for PAHs in Washington did not use total PAHs, like the fish bioassays, but instead had both total High Molecular Weight PAHs (HPAHs) and total Low Molecular Weight PAHs (LPAHs). For establishing injury thresholds for the Commencement Bay NRDA, the total HPAH results were used, which was a choice that led to lower injury estimates because the thresholds for LPAH tests were all higher (Wolotira, 2002). Using a joint plot of English sole response thresholds, at each of which percentage of exposed fish showing impacts increased and impacts themselves became more serious, and benthic invertebrate thresholds, at each of which more types of assays showed injury, four thresholds for PAHs were assigned: 20% injury at 1 ppm; 40% at 8 ppm; 60% at 17 ppm; and 80% at 70 ppm total PAHs (Wolotira, 2002). These levels were set higher than those for HCB because of known effects extending to higher trophic levels.

Once injury thresholds have been established for all SOCs, GIS assessment of the joint distribution of all SOCs over all the sedimentary benthic habitats can be used to produce an overlay of the total percent service loss arising from the joint effects of all contaminants. The combination of injuries is done multiplicatively (Wolotira, 2002). First one contaminant is assessed and the residual ecological service percentage computed. For example, if the first contaminant exists in a concentration that induces a 10% service loss, 90% of the services would be retained. Then if a second contaminant has a concentration that induces a 20% service loss, that is applied to the 90% residual, producing an additional loss of 18% more: thus the joint effect of the two SOCs acting together is to leave 72% of ecological services. Such multiplicative application prevents service losses from exceeding 100%. In addition, the computation possesses a distributive property so that the order of computation of SOC effects does not matter to the final result.

GIS was then further applied in the contaminated Hylebos Waterway under the Commencement Bay NRD case for several other purposes that are well founded and together represent a state-of-the-art assessment of injury from chronic sediment contamination (Commencement Bay Trustee Council, 2002). First, the unvegetated benthic sediments comprise not just one but several different habitats with differing levels of ecological services associated with each. So the intertidal, shallow subtidal, and deeper subtidal habitats were defined based on depths and mapped as a GIS layer. Each of these sedimentary habitats was then assigned a natural level of ecosystem services relative to a single standard, the salt marsh, which was set at unity (Commencement Bay Trustee Council, 2002). Because of the importance of intertidal flats to Chinook salmon as an important feeding ground, the intertidal sedimentary benthic habitat was assigned a pristine value of 0.9, only 10% less than that of a pristine salt marsh on the Hylebos Waterway. Shallow subtidal bottom was assigned a pristine service level of 0.7, and deeper subtidal sediments a level of 0.3, relative to 1.0 for salt marsh. This waterway also has an artificial shoreline habitat type, made of rip-rap extending along much of the industrialized zone. Mapping also included upland habitats of vegetated buffer and of upland greenbelt that extended above marshes in some places. The assignments of relative values of ecosystem services of these alternative habitats were based on literature information on ecological processes in the various habitats. The relationships between salt marsh, intertidal flat, and subtidal bottom are similar to the levels computed for those three habitats by Peterson *et al.* (2009) based upon weighted averages of documented productivities of the three lowest trophic levels. Relating ecological services of these sedimentary habitats to those of salt marshes has great utility for compensatory restoration because salt marsh is the targeted habitat for restoration (Commencement Bay Trustee Council, 2002).

After subdividing the affected bottom by habitat type, a further GIS application was used to inject landscape considerations of beneficial effects of neighboring habitats on the ecosystem services delivered by various estuarine habitats (Commencement Bay Trustee Council, 2002). The ratios of relative value of ecosystem services assigned to each unvegetated sedimentary habitat relative to salt marsh standard of unity (1.0 vs. 0.9 to 0.7 to 0.3) apply to pristine estuarine systems in which each habitat is bordered by naturally functioning habitats that achieve important interactive services. However, in the largely altered urbanized waterways, many locations lack a functioning adjacent habitat. In those conditions, downward adjustments of ecological services were made and called baseline-adjusted levels of ecological services (Commencement Bay Trustee Council, 2002). For example, any intertidal flat not associated with an adjacent functioning salt marsh or vegetated buffer was assigned a relative ecological service value of 0.75 instead of 0.90. Likewise shallow subtidal habitats lacking these vegetated connections were reduced to a level of 0.55 instead of 0.7 (Commencement Bay Trustee Council, 2002). This represents an appropriate means of handling landscape connectivity issues of known significance to habitat functioning.

The GIS layering capability was further employed to address another problem with assessing injury to urbanized, already modified water bodies, namely the presence of degradation unrelated to the contaminants at issue in the NRDA case. In the Hylebos Waterway, shading by overwater structures like piers and log rafting and wood debris accumulation were identified as important types of degradation by other stressors. Consequently, all bottom areas affected by either of these impairments were adjusted so as to be given a baseline level of ecological services of only 0.1. Then any further reduction of ecosystem services caused by chemical contamination was computed as reductions away from that lower baseline reflecting pre-existing degradation.

The final tasks involved in computing DSAYs associated with contamination of sedimentary benthic habitats relate to time. First, the current assessment of contamination levels addresses the present, but past contamination must also be inferred to know how far back the injuries extended. A case brought under CERCLA cannot claim injuries that predate the December 11, 1980 date of passage of this legislation, so that sets a maximum duration into the past. Fish contamination records for bottom-associated flatfish revealed little change in impact between the 1970s and 1980s and 1997 (Commencement Bay Trustee Council, 2002). Consequently the DSAY computation assumed present levels of contamination and ecosystem services losses from 1980 to present. Then estimates were also made of how quickly sediment remediation would lead to restoration of ecosystem services in the future. For the Hylebos Waterway, Trustees assumed that remediated sediments would increase linearly over time over a 10-year period and at that point reach full natural levels of ecosystem services. For areas left to improve under natural recovery, the Trustees assumed a 25-yr time frame to achieve full functionality. With this information, DSAYs are then computed applying the standard 3% rate to discount future services and enhance past ones.

2.2.3 Water column

The water column represents the habitat that defines a system as aquatic, either freshwater (lacustrine or riverine), brackish, or salty. Oceanic, lacustrine, and even estuarine primary production is dominated by the phytoplankton, which is comprised of single-celled aquatic microalgae. These in turn are consumed while suspended in the water by herbivorous zooplankton, including both holoplankton (species of copepods and other crustaceans that remain planktonic for their entire life) and most meroplankton (larvae of marine invertebrates that later undergo metamorphosis into a different, typically sessile adult form). In addition, where water depths are shallow, benthic suspension feeders also consume phytoplankton from out of the water column, and dead phytoplankton detritus settling to the bottom supports feeding by deposit-feeding invertebrates. Both the phytoplankton and zooplankton provide one base of estuarine and marine food chains leading to fish and shellfish, then at higher trophic levels to seabirds and marine mammals. Water quality plays a large role in ecosystem services of the water column because: (1) suspended sediments and other particles create turbidity, reducing light needed for photosynthesis; (2) dissolved nutrients are necessary for plant growth but at high levels induce nuisance algal blooms, hypoxia and other symptoms of eutrophication; and (3) dissolved or suspended toxins can reduce growth, harm reproduction, kill, and contaminate susceptible aquatic organisms. Consequently, the water column represents an extremely important habitat in any aquatic ecosystem because it is the medium in which a large fraction of the aquatic biological food chain originates.

Despite the ecological importance of water column habitat in aquatic ecosystems and its central place in defining a system as aquatic, the water column is rarely treated as a habitat in a HEA context for assessing injury from events that trigger natural resource damage assessments. We know of no examples of estimating the percent service loss of water column habitat or computing the water column DSAYs (Discounted Service Acre-Years) associated with an oil or chemical spill, despite the prevalence of this method of injury to bottom sedimentary habitats (e.g., Commencement Bay Trustee Council, 2002). Instead injuries that occur in and to the water column are expressed at the species or taxon level and thus show up as fish, shellfish, or waterbird injuries, usually quantified by production lost (e.g., NOAA *et al.*, 1999; French McCay and Rowe, 2003). For oil spills, these injuries to specific taxa or suites of functionally similar taxa are often quantified by one of two approaches. First, for especially valuable, charismatic or publicly visible animals, a common approach is collecting, sizing, weighing, and aging the dead animals by species and then using buoyancy, physical transport, and scavenging risk models to compute the fraction of all those animals killed that is included in the stranding counts (sometimes confirmed by field sampling in oiled and unoiled reference areas). This method was employed by French McCay *et al.* (2003a) to compute biomass of lobsters killed by aqueous exposure to PAHs after strong winds and wave action mixed oil spilled from the North Cape to 21 m depth in the ocean. To the biomass killed, a second component of injury is then added, based on demographic computations, to account for future production foregone by the deaths of those animals before their expected lifetime production was achieved (French McCay *et al.*, 2003a). The second method commonly used to estimate production lost from individual taxa and functional groups in response to water column exposure that induces acute mortality is to apply a modeling approach. Such models use a physical fates model of how contaminant concentration varies over space and time, validated by field measurements during and following the release. Then regional data on abundances and distributions of taxa exposed to these water column concentrations are used to estimate from available laboratory test data on acute toxicity levels the quantitative biomass lost through toxicity in the water column (French McCay and Rowe, 2003).

Several factors contribute to this typical decision not to quantify water column injuries as habitat injuries. First, oil, the most common contaminant in spill or release incidents, floats on water and does not readily dissolve into water. Consequently, even at the peak of an oil spill and even beneath floating oil, detecting PAH contamination is usually difficult and levels fall below instrument detection capacity quickly over time frames of weeks (Short and Harris, 1996a). After the 24 March 1989 Exxon Valdez oil spill of 11 million gallons, detecting any water column oil was difficult by the end of the summer (Neff and Stubblefield, 1995). Nevertheless, even when water sampling failed to reveal oil, transplanting of initially clean filter-feeding mussels that concentrate contaminants through their water filtration action revealed ongoing exposure and PAH contamination (Short and Harris, 1996b). So sampling filter-feeding invertebrates provides a more sensitive measure of injury than sampling the water-column habitat in the case of injury from oil. Second, the duration of injurious concentrations of contaminants in the water column after a release or spill is short. For oil and other contaminants, water disperses the materials widely and rapidly and helps induce degradation via exposure to light, oxygen, and physical forces. Water column injury per se is generally short in duration, provided absence of continuing injection of new contamination from reservoirs of contamination (Peterson *et al.*, 2003b). Consequently, natural water column recovery occurs rapidly, so assessment of acute mortality at the taxon level seems appropriate. Because of this short duration of injurious concentrations of contaminant levels in the water column, the time factor entering into DSAY computation would be small, leading to relatively low estimates of DSAYs despite the potential for long-lasting injuries to long-lived seabirds, marine mammals, and other species exposed through the medium of the water column. Third, attempts to assess injury to the plankton after oil spills have generally failed to show any impacts, in large part because of the wide dispersal of the contaminant and the organisms in the water column renders sampling and impact detection difficult. Fourth, the method chosen to quantify injury should lead readily to scaling of compensatory restoration. Creating new water column is not a feasible restoration option in the ocean or in estuaries so in-kind replacement of DSAYs is not an option for these marine systems. Even cleanup of existing waters as a means of creating water column service enhancement is challenging given the low concentrations of most contaminants in the water itself. However, creative and useful restoration projects targeting water quality should not be overlooked where feasible. Sedimentation and nutrient loading represent such ubiquitous challenges to water quality that projects that reduce those inputs could be valuable as restoration options.

Although long-lasting injury to the water column habitat is unlikely after spill or release incidents, permits granted for planned discharges to public trust waters can lead to persistent water quality degradation. Studies of the impacts of discharge of reactor cooling waters by the San Onofre Nuclear Generating Station (SONGS, 1989) demonstrated persistent effects on the water-column. However, even though these effects persist as long as the discharge continues, even in this case quantitative injury and compensatory mitigation were scaled in terms of injury to another habitat (kelp forest) and a group of valued animals (mid-water fishes). The ocean intake and use of water for cooling the reactors entrains plankton and fish eggs, larvae, and juveniles (SONGS, 1989). The heating and mechanical damage during passage through the plant then kills a large (and measured) fraction of the organisms that are present in the water intake. Sampling of phytoplankton and zooplankton abundances in the plant vicinity and at a control site 12 km away from the plant was unable to detect any effect of this loss of plankton. However, the death of the plankton and fish life stages resulted in enhanced concentrations of organic particles in the diffuser discharge, causing a turbidity plume that shaded a kelp forest. In addition the particulates settled out and caused anomalous sedimentation of muds onto rocky bottom inside and outside the kelp forest. Because of the shading that averaged a light reduction of 6-16% in the San Onofre kelp bed and the sedimentation that covered rock surfaces with muds and inhibited recruitment of juvenile kelps (SONGS, 1989), the area of the kelp bed declined by 80 hectares. This injury is being mitigated by creating an artificial rocky reef to serve as a kelp habitat. The details of the kelp forest injury quantification and the mitigation scaling are presented in the kelp forest sections. The injury associated with intake and mortality of fish eggs, larvae, and juveniles results in an estimated loss of 57 tons of fish annually. The California Coastal Commission did not require application of discounting and required a wetland restoration of 60 hectares, which was estimated to provide annually a biomass of adult fish equal to the annual loss through water intake mortality. Monitoring is required by the California Coastal Commission to confirm that expectations of restoration scaling are met and adaptive management will be required to meet those targets if the benefits fall short. Although this injury is being mitigated by an out-of-kind remedy, the rare salt marsh wetland is expected to produce many small fishes, matching the species and types most killed (fodder fish like anchovies especially). Consequently, even for ongoing injury by permitted discharges, no attempt was made to estimate the percentage service loss of the water column as a habitat, but instead the injuries happening in the water column were quantified by other means (a HEA for kelp habitat injury and a REA for midwater forage fish loss).

2.2.4 Oyster reef

One habitat of shallow estuarine shorelines that is at risk from oil spills, other contaminants, and shoreline accidents and is valued for its ecological services (Lenihan and Peterson, 1998; Coen *et al.*, 2007; Grabowski and Peterson, 2007) is oyster reef. Intertidal oyster reefs are especially vulnerable because floating oil can become deposited over the reef at low tide and then penetrate deep into the shell matrix and sediment-filled interstices. In addition, as a shallow-water suspension feeder, oysters filter large volumes of water (Newell *et al.*, 2007) and thereby concentrate dissolved contaminants. The high use of oyster reefs by associated benthic invertebrates and mobile fishes and crustaceans (Peterson *et al.*, 2003b) places a productive food web at risk if the reef or its oysters become contaminated. Oyster reefs are most prevalent along the mid-Atlantic, southeast Atlantic and Gulf of Mexico coasts.

Two NRD injury assessments serve as models illustrating methods for assessing injury and associated service losses in oyster reef habitat. The Lavaca Bay assessment (Lavaca Bay Trustee Council, 2000; TGLO *et al.*, 2001) provides an example of application of basic principles of injury assessment both for a complete loss of oyster reef habitat and also for persistent chemical contamination, as often associated with EPA Superfund sites. This case also demonstrates how these injury assessment methods readily allow scaling of in-kind compensatory restoration.

Another more recent assessment extended the scope of injury assessment to include the sub-lethal impacts on oyster growth (one component of secondary production) from a pulsed injection of PAH contamination from an oil spill. Because such sub-lethal effects are common, this alternative approach used in an oyster reef contamination case has generic application to injury of other habitats and species.

2.2.4.1 Complete loss of oyster reef and persistent chemical contamination

The Lavaca Bay case in Texas (Lavaca Bay Trustee Council, 2000; TGLO *et al*., 2001) involved long-term bottom contamination by mercury and PAHs discharged in wastewater from an aluminum smelter historically and other industrial activities more recently. Oyster reef habitat and associated animals suffered injury when a contaminated oyster reef was entirely removed by dredging as a component the primary restoration response actions. Oyster reef habitat had also experienced reduction in its delivery of ecosystem services over a long period of time as oysters and fishes associated with the contaminated oyster reefs accumulated both mercury and PAH contamination through their association with the contaminated bottom.

The injury to the oyster reef caused by dredging during primary restoration was assumed to have permanently eliminated those acres of oyster reef habitat and associated ecosystem services from the time of dredging forward. Computation of DSAYs (Discounted Service Acre-Years) associated with this habitat removal was straightforward because the 0.22 acres of reef was completely destroyed and the duration of the loss was assumed to be infinite, so the often challenging task of estimating the quantitative recovery trajectory over time was unnecessary.

The annual injuries to oysters and associated fish for each year of presumed historic contamination from industrial discharges were estimated as percent losses of ecosystem services for each acre of contamination, which were summed over time with discounting to yield the total DSAYs associated with contamination. A Reasonable Worst Case (RWC) approach was adopted (TGLO *et al*., 2001), which is conservative in the sense that injuries are not demonstrated but assumed to be probable on the basis of contaminant concentrations in organisms and literature evidence of the levels of those contaminants that induce negative consequences (on behavior, growth, reproduction, or mortality). Two contaminants of concern (COCs) were identified by chemical analytic analyses of tissue samples of both oysters and a representative demersal fish associated with oyster bottom (gobies): mercury as the primary COC and PAHs as the secondary COC. Sampling in the field identified the numbers of acres of oyster reef characterized by specific ranges of contamination by each COC separately.

For mercury and PAHs, literature values and the sediment quality triad approach (TGLO *et al*., 2001) were used to determine the AET (Apparent Effects Threshold). The sediment quality triad approach uses three data streams to judge whether contamination has lead to degradation of ecological services: contaminant concentrations in sediments, sediment toxicity test results, and infaunal community composition (Long and Chapman, 1985; Chapman 1990, 2000). The oyster habitat AET was based on oysters themselves, without which there would be no habitat and which comprise a large majority of the total secondary production in oyster reef habitat (Bahr and Lanier, 1981). The sediment triad analysis suggested an AET of 4.6 ppm of mercury for oysters. However, a literature review of how oyster survival, growth, and production are affected by mercury in surrounding sediments demonstrated an AET of 0.59 ppm. This lower number was chosen as the oyster habitat AET because it reflects apparent sensitivity and the assessment strategy uses a conservative (protective) approach. Then based on the actual tissue concentrations of TPAH (total PAHs), HPAH (High Molecular Weight PAHs), and LPAHs (Low Molecular Weight PAHs), the percentage service loss from PAH contamination was inferred from the review of empirical field data, which were highly variable. The probable service losses (AETs) for oysters began at 5.2 ppm of HPAHs and 17 ppm for LPAHs. Trustees then used the empirical data on type of response and magnitude of response to assign percentage service losses to oyster habitat for each COC separately. This process resulted in assigned losses of either zero or 25% because the oyster response data did not suggest any finer resolution of injury thresholds or relatively high levels of injury. To determine the joint effects of both contaminants, the two separately computed percent service loss figures were added (this can be compared to the multiplicative combination approach used in Commencement Bay, 2002, which may be more appropriate in most cases). Thus percent service losses to oyster habitat were zero, 25%, or 50%.

This same procedure was also applied to gobies, based only on expected levels of fish tissue contamination by mercury from observed field sampling of the oyster reef sediments and an assessment of the relationship between sediment mercury levels and fish tissue mercury levels. PAHs are metabolized by fish, making assessment of exposure and injury difficult. However, the mercury and PAH distributions were broadly correlated and the geographic area of mercury contamination was more extensive, so only mercury was used to scale injury. The conservative nature of the estimation for mercury provided further assurance that this exclusion of PAH injury to gobies did not result in a problematic underestimate of injury. Literature values identifying the relationships between mercury concentrations and fish behavior, growth, reproduction, and mortality were applied to assign percent service losses to resident benthic fish on oyster reef habitat. Mercury at <0.5 ppm in fish tissue is associated with sediment concentrations of <0.7 ppm and no fish injury. Mercury concentrations of 0.5-1.0 ppm in fish are associated with sediment concentrations of 0.7-1.3 ppm, with effects indicating a 10% service loss. Fish tissue concentrations of 1.0-2.0 ppm are associated with sediment concentrations of 1.7-3.3 ppm, and were assigned a service loss of 20%. Because historically available tissue sampling data for fish at this site showed that mercury concentrations were twice as high in 1981 as in 1999, percentage service losses were accordingly doubled for 1981 and a linear change was assumed between 1981 and 1999 to establish the magnitude of fish injury each year in units of DSAYs.

In the Lavaca Bay case (TGLO *et al*., 2001), the estimated DSAYS for oysters were also added to those for gobies (as a proxy indicative of all associated fishes, crabs, and other invertebrates) to determine the interim total injury to oyster reef habitat services from long-term contamination by mercury and PAHs. Total injury was constrained so as not to exceed 100% of services for any acre. A further contribution from shorebirds and wading birds that feed on oyster reef prey was contemplated. However, the method for inferring injury to birds from contamination based on contaminant levels of their prey showed that the primary COC, mercury, was everywhere less than the LOAEL (Lowest Observed Adverse Effects Level) and the NOAEL (No Observed Adverse Effects Level), so impacts to shorebirds and waders feeding around oyster reefs were assumed to be negligible.

Adding the percentage service loss from the benthic habitat (oysters) to that from the nektonic consumer trophic level (as represented by gobies, a resident benthic fish) raises a similar methodological question to that addressed by Wolotira (2002) for how to treat contributions of multiple contaminants affecting a single response variable. This raises questions of whether injuries have been double-counted. In most cases, benthic habitat can suffer loss of a certain percentage in ecosystem services while contamination can add to losses at higher trophic levels from factors not already included in habitat loss. For example, if growth, reproduction, and mortality jointly reduced oyster production by say 30%, then total production at higher trophic levels such as that represented by benthic fishes associated with oyster reef would also be expected to decline by 30% from quantitative loss of habitat. The remaining fishes could still suffer additional losses through contamination, which may affect the production of those 70% of the fish left after the habitat loss. However, some metrics applied separately to more than one trophic level may result in multiple counting.

Constraining the total service loss of an acre of habitat so that it does not exceed 100% seems appropriate for most applications, but not for all. Here in the Lavaca Bay (TGLO *et al*., 2001) case, the fish used, a goby, is closely and intimately associated with the oyster reef bottom and is resident there. Many other fishes, birds, and mammals are highly mobile. They could suffer injury that extended well beyond the geography of the contaminated habitat simply by feeding occasionally, perhaps even only once on a persistent toxic chemical from contaminated habitat as they migrate by. Thus, more migratory fish, birds, and mammals could exhibit scales of injury that go well beyond the spatial scope of the injured habitat that contaminated them. In such cases, NRDA methods may treat these species in REAs (Resource Equivalency Assessments) so that compensatory restorations could be scaled to the full scope of the resource injuries. These would be additional injuries outside those to the habitat itself, meaning that at these higher trophic levels the injured habitat could cause considerably more than 100% loss of services of higher trophic level production.

One aspect of the Lavaca Bay assessment involved translating synthesized data on how chemical concentrations affect oyster response variables of concern, such as growth, reproduction, and survival, to percent service loss. An alternative approach to that used in Lavaca Bay would be the use of simple demographic modeling to synthesize effects on growth, reproduction, and survival into a single metric of lost oyster production. This process would involve similar or even identical data syntheses to determine how ranges of contaminant concentrations relate to oyster growth, reproduction, and survival from literature information. Thus instead of assigning a percent service loss to each concentration range, that percentage would be estimated by demographic life-table computations. Such computations are illustrated for different purposes but applied to oysters in the Chalk Point case restoration scaling for oyster reef restoration (French McCay *et al*., 2002). The demographic approach involves estimation of recruitment rate of 0-yr-olds per unit area of oyster reef, and application of age-specific survivorship in a Leslie matrix to project how those numbers change over the years until all are dead. This process produces the data which can be combined with size-at-age information to compute total oyster production under conditions of attaining a stable age distribution. This allows computation of total production. Then any decrease in recruitment, growth rate, or survival from contamination can be readily computed, thereby allowing conversion to percent decline in secondary production of oysters. This number could replace the somewhat subjective estimate used in Lavaca Bay

(TGLO *et al.,* 2001). This more objective method requires more effort and perhaps site-specific empirical information on recruitment rate, growth rate, and age-specific survivorship of oysters.

2.2.4.2 Quantifying production losses from sublethal impacts of spilled oil

Recent developments provide a method to estimate the quantitative sublethal injury induced by contaminant uptake by oysters after a spill. Three degrees of oil exposure were estimated from visual surface oiling in the week following a spill in an east coast harbor of the United States and the coverage estimates of oyster reef habitat arose from systematic mapping of the extent of oyster reef oiling at each intensity. While no evident mortality occurred, sampling of PAH levels in oyster tissues demonstrated elevated levels that corresponded to the three levels of oiling. Chemical analyses were applied to demonstrate the petrogenic source and remove pyrogenic sources from inclusion in estimating contamination impacts. Elevation in PAH concentrations in oyster tissues above background levels was therefore used as a proxy for effects on ecosystem services from affected oyster reefs.

To use the PAH concentrations in oyster tissues about 1 week after the spill as an index of exposure to oil required baseline PAH contamination evidence. Repeat sampling 6 months after the spill was used to provide baseline PAH tissue contamination for 7 oiled sites. The use of these later data as baseline presumes that PAH depuration would have occurred within 6 months. Any oyster reef that failed to exhibit PAH concentrations in oyster tissue that were elevated over this background was excluded from injured reef area.

The metric used to quantify the loss of oyster ecosystem services was the estimated percent decline in normally expected oyster growth. Based on extensive scientific research on the physiological and ecological responses of mussels, oysters, and other large suspension-feeding bivalve mollusks to PAHs, the tissue concentrations of PAHs can be linked to expected percentage suppression in growth rate, using scope-for-growth information (Bayne and Widdows, 1978). Individual growth is one component of production: reducing average growth rate of oysters by 50% translates directly into a 50% reduction in secondary production of oysters, a common proxy of ecosystem services (e.g., Peterson *et al.*, 2003b). Consequently, to compute DSAYs associated with the sublethal impacts of PAH contamination affecting oyster growth rate, the magnitude of ecological service loss was inferred for each level of oiling from percent declines in growth rate, presumed duration of lower growth (6 months or half a year), numbers of reef acres with that level of oiling, and discounting. Thus, this method of assessing injury to oyster reef habitat from contamination by PAHs demonstrated (1) exposure and injury in the form of tissue contamination that is known to reduce scope-for-growth; (2) geographic coverage of each level of contamination; (3) probable duration of the sublethal depression in growth; and (4) scaling computations using discounting from the time of injury until when restoration would become complete. This scope-for-growth method for injury estimation is limited to those few well-studied suspension-feeding bivalves (mussels and oysters) and limited to contamination by PAHs. Consequently, it may have more limited applicability compared with the AET method applied to Lavaca Bay. In addition, this approach is relevant only to sub-lethal impacts so any more serious spill that induces mortality of the oysters or mussels or influences higher trophic levels would require additional methods of injury assessment even if reduced production from lower growth of oysters or mussels were used as one contribution to injury.

2.2.5 Mangroves

Mangal habitat is found in tropical and sub-tropical latitudes where the minimum monthly air (Chapman, 1976) and water (Duke *et al.*, 1998) temperatures are 20^{o}C. In the mainland U.S., mangroves are now restricted to south Florida but are expanding northwards as climate warms. Mangrove forests occur on Puerto Rico, the U.S. Virgin Islands, and Pacific Ocean protectorates. Mangrove forests are collectively comprised of an assemblage of salt-tolerant trees and shrubs that colonize low-energy depositional environments and waterlogged, oxygen-deficient, saline soils within the tropics (Cintrón-Molero, 1987). Mangrove habitat replaces salt marsh in the tropics, although the functional roles are not necessarily identical. Like their salt marsh counterparts at higher latitudes, mangrove forests contribute to estuarine and coastal oceanic food chains through production and export of organic materials, provision of biogenic, structured habitat for fishes, crabs, and many birds and mammals, and biogeochemical cycling (Cintrón-Molero, 1987). Mangrove roots, branches, and pneumatophores provide nursery and spawning habitat for wildlife, fish, and shellfish, as well as nesting, roosting, and foraging sites for birds. Mangrove inhabitants in south Florida include various invertebrates (e.g., sponges, crabs, tunicates, bivalves, and spiny lobsters) and fishes (e.g., bluestriped grunt, gray snapper, dog snapper, and common snook) (Ellison and Farnsworth, 2001). Sesarmid, occypodid, and portunid crabs are especially characteristic of mangal habitats, often grazing a large fraction of leaf production and serving to support crab harvests in many parts of the world. Mangroves aid in the prevention of coastal erosion by acting as a physical buffer against impacts of major storm events. Additionally, mangroves filter nutrients, sediments, and other pollutants carried in terrestrial runoff, thereby enhancing water quality of the estuary and coastal ocean. Mangal habitat is extremely depleted and threatened worldwide by forestry, coastal development, and conversion into aquaculture ponds.

Records of oil spills impacting mangrove habitats go back into the 1960s. Two cases, the 1993 Tampa Bay oil spill and the 1986 Panama Oil Spill (Bahía Las Minas), appear to offer some guidance on how injury was quantified and compensatory restoration was scaled.

2.2.5.1 Mangrove injuries from oiling

The 1993 Tampa Bay Oil Spill involved the collision of two barges near the mouth of Tampa Bay, Florida. Jet fuel (type A), diesel, and gasoline were discharged into the Bay and the oil slick was later dispersed along the shorelines of adjacent counties, notably in Boca Ciega Bay. Many types of habitats and natural resources were negatively impacted by this event; but we restrict our presentation to injury assessment in the mangrove habitat.

The trustees combined field measurements from ground and aerial surveys over multiple years, a literature review, and technical expertise and judgment to estimate injury and express it in units of percentage loss of ecosystem services. Injury assessment revealed that some mortality of adult trees occurred as well as limb loss and foliage decline. However, earlier life stages, seedlings and juveniles, plus planted seeds, exhibited more sensitivity to the oil. Trees species varied in susceptibility. Sediment oiling persisted with little evidence of increased weathering over time, implying long-term loss of services related to soil condition. No evident assessment of effects on mangrove crabs is presented in the documentation of injury, perhaps an omission given the susceptibility of crustaceans to toxic compounds including PAHs (FDEP *et al.*, 1997). The acres of oiled mangal habitat totaled 14.4, with 9.2 acres lightly oiled, 4.3 acres moderately oiled, and 0.9 acres heavily oiled. Resources service losses were reportedly computed based upon observations of: (1) changes in mangal species composition and structure (age class); (2) survivorship (by species and age class); (3) oil effects and persistence on and in mangal sediments (% of visible and buried oil in sediments over time); (4) changes in faunal use of mangal habitat; and (5) damage caused by clean-up efforts. Interim loss of ecosystem services was reportedly based upon area, degree and duration of oil exposure, species and age class of mangrove plants, and specification of ecosystem services lost. The documents available for this case, however, reported only the general HEA principles and not the explicit procedure used to quantify injury and scale compensatory restoration.

2.2.5.2 Mangrove injury assessment with historical injury data

Rarely do trustees have a well-defined site-specific baseline on which to assess quantitative habitat or resource injury by incorporating before-after contrasts. The damage assessment from the 1986 Refinería Panamá oil spill was unique in that many of the coastal habitats in Bahía Las Minas region had been monitored and investigated in great detail by the Galeta Marine Laboratory (Smithsonian Tropical Research Institute) as a consequence of a 1968 oil spill. Results from comprehensive time-series monitoring of local biota and the physical environment had been published (Macintyre and Glynn, 1976; Cubit *et al.,* 1986; Cubit *et al*., 1989) providing quantitative measures of ecosystem status and its natural variation prior to the 1986 spill. The 1986 spill was caused by an oil tank rupture at an oil refinery, spilling at least 8 million liters of medium-light crude. On-shore winds initially restricted oil to the tropical shoreline adjacent to the refinery for 6 days before rains flushed the slick off shore.

Aerial photographs taken before and after each oil spill, 1966 and 1973 for the 1968 spill and 1979 and 1990 for the (December) 1986 spill, were used to create maps that delineated oiled mangrove forest and other habitat types. Habitat was then classified into six categories: (1) deforested areas after the 1968 spill and after the 1986 spill; (2) mature mangrove forests with normal canopies; (3) mature forests with open canopies in 1990 that had been closed in earlier photographs); (4) areas that appeared unaffected by either spill; (5) mangrove areas newly formed after 1966; and (6) uplands (Duke *et al*., 1997). Ground surveys were used to support the above classifications. Mangrove forest health was ascertained by an assessment of mangrove tree parameters (stem height, girth, and density) and an “angle gauge” technique (Cintrón and Schaeffer-Novelli, 1984) for older stands. Total above-ground biomass of each tree was estimated using an allometric equation derived by Cintrón and Schaeffer-Novelli (1984) for red mangrove trees:

$$\text{Total Biomass} = 23.6398\,(D^2H)^{0.5902}$$

where D is stem diameter (cm) and H is stem height (m). The time-series of aerial photographs revealed an area of devegetated mangrove forest that became replaced by the Caribbean Sea through subsequent shoreline erosion. Since natural recovery of mangroves was not possible for this acreage, full compensation for the permanent loss of this habitat area was calculated.

The methodology used in this case assumed that a total loss of ecosystem function had occurred within deforested mangrove habitat and that sublethal damage caused by the spill was indicated by vegetation exhibiting an open canopy. Quantification of sublethal damage of mobile organisms that occupy mangrove habitat could be determined and scaled in a manner similar to that of marsh habitat (as in the muskrat REA assessment in Chalk Point (NOAA *et al.*, 2002). Based on aerial photographs, most sites that were deforested after the 1968 spill had become densely covered with young recruits by 1973 through natural recovery (Duke *et al.*, 1997). This recovery was confirmed by field studies conducted in 1989 and 1992. A similar pattern of recovery was expected for the deforested mangrove forest areas that resulted from the 1986 oil spill. For the purposes of scaling compensatory restoration, these observations of the extent of mangrove habitat lost, the degree of damage and the expected rate of recovery at each level of injury was quantitatively estimated.

2.2.6 Seagrass

Seagrasses are clonal marine flowering plants that occur in shallow soft-sediment habitats along the shores of estuaries throughout most of the world (Williams and Heck, 2001). The 12 genera and approximately 60 species of known seagrasses are found at water depths ranging from the intertidal zone to as deep as 50 m (Fonseca *et al*., 2002). In addition to the high primary production of seagrasses with associated epiphytes and microphytobenthos, seagrass habitats present three-dimensional emergent structure providing refuges from predation, recruitment sites, and nursery function (Williams and Heck, 2001). The seagrass leaf canopy baffles the flow of water, and along with the rhizome and root mat stabilizes sediments, filters the water column and recycles nutrients between the sediments and water column (Fonseca, 1996). Globally, seagrass habitat abundance has decreased dramatically in recent years (Roblee *et al.,* 1991; Hall *et al*., 1999; Seddon *et al*., 2000; Azzoni *et al*., 2001; Plus *et al*., 2003) due to stressors such as eutrophication and locally increased turbidity and salinity, as well as the direct impacts of vessel groundings, bottom-disturbing fishing practices, and other anthropogenic disturbances (Livingston, 1987; Short and Wyllie-Echeverria, 1996; Halpern *et al*., 2007).

Much of our understanding of quantification methods of seagrass ecological services, service losses, and restoration benefits has been derived from experiences in the Florida Keys National Marine Sanctuary. NOAA and the State of Florida have served as trustees for many cases in which seagrass habitat has been destroyed or injured. Approximately 650 vessel groundings are reported in the Florida Keys National Marine Sanctuary each year, with cumulative effects amounting to over 30,000 acres of scarred sea grass habitat (Kirsch *et al*., 2005). The appearance of such dramatic impacts as a result of numerous and often minor incidents led officials to develop a systematic approach to injury assessment and restoration scaling that cost-effectively evaluates small-scale losses. This expedited damage assessment and restoration process is described in detail by Kirsh *et al.* (2005). It is worth noting that assessment efforts for sea grass in the Florida Keys have led to two court cases in which HEA methods were upheld: *U.S. v. Fisher,* 22 F. 3d 262 (11^{th} Cir. 1997) and *U.S. v. Great Lakes Dredge and Dock,* 259 F. 3d 1300 (11^{th} Cir. 2001). This assessment protocol is known as the “Mini-312” Program, so named for its reference to Section 312 of the National Marine Sanctuaries Act (for more information visit http://www.darrp.noaa.gov/partner/mini312/protocol.html). The Mini-312 program offers a standardized methodology for the quantifying of service losses in seagrass habitat and is described below.

2.2.6.1 Direct seagrass injury from vessel groundings

Injuries to seagrass habitats from vessel groundings generally take one or more of the following forms: (1) propscars - narrow, linear denuded features that result from vessel's propeller passing through seagrass bed (Hammerstrom *et al*., 2007; Zieman, 1976 as presented in Kirsh *et al*., 2005); (2) blowholes - areas of excavated seagrass that result from accelerating the vessel engine in an attempt to refloat it (Whitfield *et al*., 2002 as presented in Kirsh *et al*., 2005); and (3) berms - areas of buried seagrass created by deposition of sediments excavated from blowholes (Duarte *et al*., 1997 as presented in Kirsh *et al*., 2005). Research has indicated that injuries of these kinds can adversely affect habitat services (Bell *et al*., 2002; Whitfield *et al*., 2002; Uhrin and Holmquist, 2003). Removal of berms, planting of sea grass shoots, and addition of nutrients in the injured habitat have been shown to accelerate habitat recovery (Fonseca *et al*., 2002). The party responsible for a grounding incident must pay for these onsite primary restoration actions, but additional restoration is also required to compensate for interim losses. Application of the Mini-312 Program protocol involves: (1) determination of the spatial extent of the injury; (2) characterization of both injured and adjacent reference habitat areas; (3) development of a spatial seagrass recovery model used to estimate the recovery trajectory of the injury; and (4) application of a habitat equivalency analysis (HEA) to quantify the interim lost resource services and determine the scale of compensatory restoration.

Scaling the restoration to grounding-related seagrass injuries requires an estimate of habitat services lost. In the Florida Keys sanctuary, the metric used for habitat services is above- and below-ground biomass of the seagrass plants. This metric is relatively easy to assess and corresponds closely to plant cover, which plays a crucial role in the provision of food and shelter for marine organisms and is important for nutrient cycling and sediment stabilization (Williams and Heck, 2001). Above-ground biomass is assessed by counting the number of seagrass shoots per square meter in a random selection of 0.25-m^2 quadrats placed on the sea floor. The number of shoots is considered a proxy for above-ground biomass. Below-ground biomass is determined using literature-based, species-specific ratios of below-ground to above-ground biomass. When more than one species of seagrass is present, the calculations are performed separately by species and then added together. Although the biomass proxy is not denominated in any specific units, this does not matter because the objective is only to calculate relative changes in habitat services in percentage terms. The initial percent service loss at the injury site is calculated by dividing the biomass density at the injury site by the biomass density in the surrounding uninjured habitat. The recovery path at the injury site is determined using a computer model (described in Kirsch *et al*., 2005) that accounts for the growth rates of the species present at the site and the size, type and spatial pattern of the propeller scar, blow hole or berm. The model assumes that recovery enhancements will be undertaken in the impacted area as part of the primary restoration.

The detailed process of the Mini-312 Program involves a sequence of steps. First, field observations of injury and bathymetric measurements are used to determine the spatial (including volumetric) extent of the injury. These data are then imported into a GIS program for mapping. Aerial photography may be employed when the spatial extent of injuries is large and geometrically complicated. Second, a modified Braun-Blanquet method (Braun-Blanquet, 1932; Kenworthy and Schwarzchild, 1998; Fourqurean *et al*., 2001) can be used to efficiently characterize both injured and reference (adjacent, undisturbed) habitat areas by quantifying the composition, percentage cover, and above-ground density of each seagrass species. This method involves determining species presence and a cover abundance value within replicate 0.25-m^2 quadrats. The cover abundance scale values for seagrasses and other species are: 0 = not present, 0.1 = solitary specimen, 0.5 = few, with small cover, 1 = numerous, but less than 5 percent cover, 2 = 5-25 percent cover, 3 = 25-50 percent cover, 4 = 50-75 percent cover, 5 = 75-100 percent cover. To determine the percentage cover for seagrass, the Braun-Blanquet scores, converted to percentages (using range midpoints), are averaged over all of the quadrats placed within each feature (propscars, blowholes, and berms). Kirsh *et al.* (2005) suggest that for purposes of interpolation, a Braun-Blanquet score of 1 is then equivalent to 3% (midpoint of 1-5%) cover, 2 is then equivalent to 15% (midpoint of 5-25%) cover, and so forth; in cases where a Braun-Blanquet score is 0 < 1%, a percent cover of 1% is given. The loss of percentage cover of seagrass as a result of the grounding can then be assessed by comparing the percentage cover of the injured area to that of the undisturbed area. Third, the seagrass recovery model is a spatially explicit, cellular automata modeling technique based upon a quadratic equation that predicts the recovery time of a given feature, in which recovery occurs only via in-growth from the periphery. Recolonization rates are determined from literature review and empirical assessments (Fonseca *et al*., 2004; Hammerstrom *et al*., 2007) and are specific to both species and geography (Fonseca *et al*., 2000). Fourth, the above- and below-ground biomass of seagrass is the resource metric used for HEAs in the Mini-312 program, as this metric is easy to assess and serves well as a proxy for the ecosystem services that seagrass habitats provide. In addition, this metric allows HEA to capture differences in the quality of services provided by injured versus replacement habitat or resources (Fonseca *et al*., 2002). Field assessment (step 2) has determined the above-ground percent cover of each seagrass species in both the injured and adjacent reference areas, thus allowing the percentage of above-ground biomass loss to be calculated from the difference in seagrass cover. Below-ground biomass is also estimated from the field-measured above-ground biomass loss and the known below-to-above-ground biomass ratios for each species (Kirsh *et al*., 2005) (see Table 2.7). Summing the m^2-years of seagrass losses over all seagrass species until primary restoration has achieved recovery provides the total estimate of injury after application of discounting.

Table 2.7. Geographic ranges and below-to-aboveground biomass ratios of selected seagrass species

Seagrass species	geographical range	below-to-above-ground biomass ratio	reference
Halodule wrightii	temperate & sub-tropical	1.669	Kirsh *et al*. 2005
Ruppia maritima	temperate & sub-tropical	Varies seasonally; mean= 0.25	Cho & Poirrier 2005
Syringodium filiforme	sub-tropical	0.651	Kirsh *et al*. 2005
Thalassia testudinum	sub-tropical	3.144	Kirsh *et al*. 2005
Zostera marina	temperate	Varies seasonally; 1230.0 – 45.0	Holmer and Laursen 2002

2.2.7 Coral reef

Coral reef habitat and its associated ecosystem is among the most threatened in the world from physical habitat destruction, overfishing, pollution, and global climate change – especially ocean warming and consequent coral bleaching and disease outbreaks plus ocean acidification, which dissolves the calcium carbonate reef matrix (Bruno and Selig, 2007). Coral reefs are the hot spot of ocean biodiversity, providing many important and unique ecosystem services (Knowlton and Jackson, 2001). Coral reefs support socially important food fisheries and a valuable aquarium fish trade. Nevertheless, the fish probably have higher value if not harvested and left in the water because of their appeal to tourists and their support of an international tourism industry worldwide. The proximity of reefs to often low-lying atolls and sandy shores provides a structural breakwater protecting against severe cyclone damage. Many organisms on the coral reef are chemically rich, offering bioactive natural chemical products of value to the pharmaceutical industry. Coral reef ecosystems are also among the most productive in the world despite their location in oceanic regions of low natural productivity (IUCN/UNEP, 1985; Hubbell, 1997; Clark, 2002), supporting communities of hard and soft corals, fish, and invertebrates that are inter-related through many complex associations. Scientific study of how coral reefs are organized provides important generic insight into evolutionary and ecological processes. The economic and environmental benefits of coral reefs to the Caribbean-Gulf of Mexico region have been estimated at $375 billion a year (NOAA, 2006).

Coral reef ecosystems occur in shallow waters of the tropics and subtropics, although deep-water corals have been recently discovered in many cold regions like the Aleutian Islands and the Gulf of Maine. Within US waters, coral reef habitat can be found in the Caribbean-Gulf of Mexico (south Florida, Puerto Rico and U.S. Virgin Islands) and the central Pacific (Hawaiian Islands, Guam, American Samoa and small pockets in the Central Pacific Basin) (Pew Oceans Commission, 2003; US COP, 2004). Much of the history of quantitative injury assessment and compensatory restoration on coral reefs has followed vessel groundings within the Florida Keys National Marine Sanctuary, where boating is so prevalent. Frequent coral reef injuries from sedimentation associated with beach nourishment (*e.g.*, Lindeman and Snyder, 1999) rarely elicit quantitative damage assessments and compensatory restoration.

Documents reporting on injury assessments to coral reef habitat serve to provide an overview of the process. However, we were unable to identify a case study that exemplifies a specific model approach to coral reef injury assessment for any type of damage. Instead, this section summarizes the NOAA Coral 312 Program, in which procedural steps of coral damage assessment are discussed, and discusses the *Berman* oil spill and vessel grounding case (Tetra Tech, 2006).

2.2.7.1 The Coral 312 Program

The NOAA Damage Assessment, Remediation, and Restoration Program (DARRP) developed a program for assessing and quantifying damage to coral reef habitat, the Coral 312 Program (http://www.darrp.noaa.gov/partner/coral/damage.html). The protocol involves three key steps: (1) field assessment of injury; (2) determination of recovery trajectory; and (3) application of an ecological services scaling model. First, in the injury assessment stage, the injury site is delineated and mapped, and the extent of the injury is depicted both quantitatively and qualitatively, using field sampling transects for small-scale injury and aerial photos for injuries that are large and geometrically complicated. Both injured and nearby reference coral reef habitat is characterized and documented using video and photographs, which are then digitally rectified to create a digital map of the overall site. Photo quadrats may be used to estimate percent cover and relative abundances of corals and other large epibiotic species. Second, the expected recovery trajectory is determined by a literature review and best professional judgment. Critical inputs to this step include species type, degree of injury, primary restoration efforts and implications, as well as the overall environmental condition of area. Third, HEA is the approach of choice to develop an ecological services scaling model in which information on injury obtained from the first two steps is incorporated into a model that matches interim losses of public trust resources with compensatory restoration.

The actual recovery of injured coral reef ecosystems is dependent on both the geometry of the injury and the realized recruitment patterns of the various coral species. Whitfield *et al.* (2002) developed a recovery simulation model that combines the use of a probabilistic technique using SAS7 and a deterministic approach using ArcINFO7 to create an iterative process by which recovery is predicted. On a grid map depicting the spatial pattern of the injuries, the status of any given cell is based upon two main factors: (1) the state (percent cover and species types) of nearest coral neighbors; and (2) a probability function that applies different growth rates and propagule recruitment probabilities to different coral species. This model calculates the overall recovery trajectory based upon the cumulative percent recovery contributed by all coral species. Whitfield *et al.* (2002) emphasize that the accuracy of the predictions is limited by the quality of available input data. Better information on species-specific recruitment and horizontal and vertical growth rates would enhance model accuracy.

2.2.7.2 Coral reef injury assessment

In 1994, the *M. J. Berman* went aground on a coral reef near San Juan, Puerto Rico (Tetra Tech, 2006). Because stormy weather prevented rapid access, the barge remained on the reef for 8 days, releasing over 800,000 gallons of No. 6 fuel oil into nearshore waters and repeatedly causing physical damage to the coral reef on which it sat. In determining compensation for lost ecological services of coral reef habitat, the *Berman* case trustees classified damaged reef by level of physical injury: (1) severely impacted (structural damage and destruction), for which 80 % of services were judged lost; and (2) partially impacted reef (scouring and breakage of encrusting organisms), for which 40 % of services were judged lost. The complete recovery of the severely impacted and partially impacted habitat was estimated to require 50 and 30 years, respectively, and a linear recovery trajectory was assumed. The levels of coral reef injury were determined using underwater diver surveys of both the damaged coral reef and adjacent unaffected (reference) coral reef, based on a standardized methodology from Causey (1990), in which structural damage is assumed to relate directly to loss of ecosystem productivity and function. The total two-dimensional area of damaged coral reef in each injury class (4,600 m^2 of severe and 4,700 m^2 of partial injury) was multiplied by the corresponding percent loss in services and the two added to determine the area of fully functional coral reef for which ecosystem services were to be compensated. Additional reef habitat of 1,000 m^2 was injured by loose rock and rubble during the grounding and subsequent removal of the vessel; however, this damage was not included in injury computations and thus went uncompensated. A primary restoration project targeted stabilization of this rubble zone to prevent further reef injury.

2.2.8 Kelp Forest

Like other biogenic habitat, kelp forests on shallow rocky reefs support dramatically enhanced biodiversity and density of benthic invertebrates and mobile fishes and crustaceans (Witman and Dayton, 2001). Kelp forests are restricted to the shallow rocky subtidal zones of temperate coastlines. Kelps, large brown algae in the orders Laminariales and Fucales, are constrained to the water depths extending down to 10-25 m, depending on light penetration. Faunal zonation with depth is characteristic of kelp forests and processes of competition, predation, and physical disturbance all play roles in driving dynamics and pattern of benthic communities within kelp forests (Mann and Lazier, 1996; Witman and Dayton, 2001). By projecting into the water column, kelps baffle currents, creating less energetic environments for associated animals and inducing deposition of suspended particles, including larvae of benthic invertebrates. The kelps themselves provide surface for attachment by epiphytes and benthic invertebrates as well as a food source for grazers like sea urchins, large gastropods, and numerous crustaceans. Kelp forests harbor several species of fish and invertebrates that are targets of commercial and recreational fisheries (Dayton *et al.*, 1998).

2.2.8.1 Injury to kelp forest habitat and resources

Kelps are sensitive to variation in turbidity, which influences light penetration through the water column, and to sedimentation, which can cover the rock surfaces and reduce their suitability for attachment by kelp holdfasts. Injury to kelp forests and associated faunal resources can be induced by operation of power plants that utilize ocean water as a coolant and discharge those waters back into the sea. This process of heating the water and passing it through turbines and cooling systems kills the planktonic and small nektonic organisms that are entrained into the seawater intake, such that the discharge includes high loads of particulate detritus, causing both shading and organic deposition on the bottom in any nearby kelp forest habitat down current (SONGS, 1989). Because permits are required before construction of such large industrial operations environmental impacts can be assessed quantitatively using a pre-planned design to serve to develop compensatory mitigation required as part of the permit. This capability for conducting planned assessments of injury associated with need to mitigate for permitted projects distinguishes compensatory mitigation from almost all compensatory restoration. The impact study conducted to evaluate the injury caused by bringing on line units 2 and 3 of the San Onofre Nuclear Generating Station resulted in the development of a powerful statistical assessment design for testing ecological impacts, the BACI (Before-After Control-Impact) design (Stewart-Oaten *et al*., 1986).

This BACI statistical approach to quantifying impacts involves monitoring the abundance, density, or concentration of the target response variable of interest (for example, bottom area covered by a kelp forest) at the “impact” site and at a paired “control” site during a sufficiently long time period “before” construction and then subsequently “after” construction (Stewart-Oaten *et al*., 1986). A BACI design has logical advantage in the rigor of its inference over alternative designs. Specifically, a before-after contrast at an impact site must assume that no temporal change in the response variable would have occurred in the absence of the construction activity. An alternative method of comparing the impact site to one or more control sites after construction must assume that only minimal pre-existing spatial differences existed and that observed differences are a consequence of construction. The use of a BACI overcomes need for these assumptions because spatial differences are controlled for by establishing the relationship between the putative impact and control sites before the project begins and natural temporal change is controlled for by using dynamics of the control site from before to after project construction. The BACI test then becomes a test of the interaction, asking whether the pre-existing difference between the putative impact and control sites changed after project construction. This methods requires advance knowledge of the location of the impact (and advance monitoring of impact and control sites), so the application of BACI approaches is almost always restricted to pre-planned assessments of planned construction projects that receive permits. Thus its use applies to compensatory mitigation.

In the assessment of impacts of the discharge of cooling waters by SONGS, the Marine Review Committee chose to evaluate impacts of area of a nearby kelp forest, density of large marine invertebrates on the bottom under the kelp canopy, and density of kelp fishes (SONGS, 1989). Physical measurements on site demonstrated the scope of likely impacts of discharges through the full suite of oceanographic conditions that impinge upon the site. Seawater taken into the powerplant is warmed by 19 degrees C. The discharge plume of heated water containing enhanced sestonic particulates from dead organisms is diluted rapidly, with dye measurements showing about 0.5% of initial concentrations at 7 km upcoast and 11 km downcoast. The average reduction in light intensity in the kelp forest was 25% on days when the currents moved downcoast toward the bed and the reduction averaged 6-16% over the year. Finally, field measurements also showed a 48% increase in particulate deposition on the floor of the kelp bed at 400 m compared to 1400 m from the discharge diffusers. This physical characterization provides indication of the spatial scope and intensity of impacts and helps confirm the mechanisms so as to make conclusions from monitoring more compelling.

The quantitative extent of injury to the kelp habitat at the San Onofre kelp bed and its large benthic invertebrates and fishes was largely assessed by the BACI approach using monitoring survey data from before and after start-up of SONGS Units 2 and 3 as compared to analogous temporally paired monitoring data at the control San Mateo kelp bed (SONGS, 1989). Relative to before-after changes at the San Mateo control kelp bed, the San Onofre kelp bed had 60% less bottom area covered by moderate and high densities of kelp, or 80 hectares. This then represents the quantitative injury to kelp forest habitat from operation of Units 2 and 3 of SONGS. Several methods were employed in separate assessments of kelp abundance and areal cover, but the method that was most valuable for conducting the BACI analysis because of its temporal duration was one in which kelps were counted four times annually on fixed transects in both kelp beds (SONGS, 1989). Down-looking and side-scan sonar surveys were also deployed to provide additional data sets on coverage of moderate- and high-density kelps. Surveys were also conducted in which numbers of newly recruiting kelp plants were counted on bottom quadrats. The evidence of reduced light levels in the San Onofre kelp bed after beginning of operations of Units 2 and 3 and the greater downward flux and deposition of sestonic particles provided confirmation of a mechanistic connection between operation of the cooling system and kelp habitat loss. A reduction in recruitment to the adult population rather than higher mortality of existing adults explained most of the loss of kelps.

Kelp forest fishes were sampled by visual surveys made by divers, counting in two "before" and two "after" years, the numbers of fishes by species in bottom transects and in water column transects in both the San Onofre and the San Mateo kelp beds. Bottom fish are associated with kelp presence but also occur on rocky bottom lacking kelp, whereas the water-column fishes only occur inside kelp canopy (SONGS, 1989). Sampling was stratified by distance from the SONGS diffusers, with two stations (close and far) in the impact and in the control kelp bed. Then sampling was stratified by areas with and without (defined as less than 4 kelp plants per 100 m^2, which is the limit detectable by side-scan Sonar) kelp plants, so as to be able to determine if fish declines were solely a consequence of declining area of kelp habitat or also represented changes within the kelp habitat. This sampling and subsequent analyses demonstrated that bottom fish declined significantly by BACI analysis in total abundance by 70% and in biomass by 73% in the San Onofre kelp bed. This decline appeared to be in part a consequence of the 70% decline in area of kelp habitat in the SONGS kelp bed during those sampling years relative to the changes in the control San Mateo kelp bed. Mid-water fishes exhibited a relative decline in abundance of 17% and in biomass by 33%, but neither was statistically significant (SONGS, 1989).

Large benthic invertebrates like abalone, sea urchins, starfish, snails, sea cucumbers, and clams were sampled in a BACI design through visual counts made by divers. A uniform grid of quadrats was sampled at each station to cover the largest possible area. At the San Onofre kelp bed two impact stations were used, one near (500 m away from) the diffusers and the other far (1500 m away). The control site at the San Mateo kelp bed was sampled using just a single station. Water depth was set equal at all stations (14 m). Sampling occurred in a temporally paired design on 10 dates before and 8 after operations of Units 2 and 3 began. Divers tallied the abundances of 37 species of large invertebrates, 20 of which ultimately proved to be common enough for separate statistical analysis. These rarer species were incorporated into statistical analyses through identification of 5 pooled groups of species: all snail, muricid snails, non-muricid snails, sea urchins, and sessile invertebrates. Larger invertebrates were selected because they could be reliably detected even when water turbidity was relatively high and because this group includes species like sea urchins and seastars that can have important community-wide impacts on kelp forest communities. Separate BACI analyses were conducted for sampling results of the near and far impact stations. Because another study had demonstrated that SONGS operation induced substantial fine sedimentation on the bottom in several areas of the impact kelp bed and that this fine sediment persisted covering rock surfaces, divers also recorded the presence of sedimentation on each quadrat so that analyses could also assess the potential response of the large benthic invertebrates to SONGS-related fine sedimentation.

Results of the BACI analyses demonstrated significant declines in about two thirds of the species or groups tested, of which all but one reflected a decline at the impact kelp bed (SONGS, 1989). The magnitude of the declines was greater in the near impact station at about 80% of expected abundance as compared to 60% and the far impact station in the San Onofre kelp bed. The only species that increased significantly in the impact kelp bed was a deposit feeder, a sea cucumber, that may have benefited from the organic sediments as food. Snails were the species most negatively impacted, probably because of their close association with rocky substratum. Densities of large invertebrates were significantly lower on quadrats that showed sedimentation, but this did not explain the full magnitude of reductions in large invertebrates.

The quantification of injuries to the kelp habitat itself in hectares lost and to fishes in biomass reduction and large benthic invertebrates in abundance reduction can be combined to scale compensatory mitigation through construction of an artificial reef for kelp forest establishment (Ambrose, 1990). These three major impacts can each be mitigated and the spatial scale of the reef creation determined so that it is expected to replace each category of loss. Monitoring can determine whether expected replacements have occurred and dictate adaptive changes in the reef if mitigation falls short. Mitigation benefits are quantified in the units of injury: area of moderate- to high-density kelp forest, abundance of large benthic invertebrates, and biomass of fishes.

2.2.9 Other habitats

Other habitats suffer injury from contamination or from some other anthropogenic disturbance with a degree of regularity that implies a possible demand for HEA analysis to provide compensation. Oiling of sand beaches is common and often very heavy. Examples include the Santa Barbara oil spill in 1989, the North Cape oil spill near Point Judith in Rhode Island, and the Fort Lauderdale Mystery Spill (NOAA and FDEP, 2002). The ocean beach is an exceptionally productive habitat, with wave energy enhancing flux of diatoms and other microalgae to dense populations of intertidal filter-feeding invertebrates. These include mole crabs, beach clams, amphipods, and various polychaetes. In addition, where kelp forests exist off-shore, kelp wrack on beaches supports high arthropod production. These invertebrates in turn support shorebirds, surf fishes, and crabs. Many surf clams and surf fishes have commercial and recreational value in fisheries. Several of the shorebirds, especially including plovers, are listed species of conservation concern. Threatened and endangered sea turtles use the sand beach for egg laying and embryo development. Consequently, the sand beach habitat has substantial public trust value associated with its ecosystem services. Most compensatory restoration for sand beach injuries is achieved via REAs in which the scaling is done on the basis of one or more resources of recognized value. Such approaches are likely to omit large losses of secondary production by benthic invertebrates of wide-spread food web importance. However, most such depressions in beach invertebrate populations are probably short-lasting, except where lasting modifications of the sedimentary habitat are induced, such as replacing beach sands with coarser materials like mine tailings (McLachlan, 1996; Peterson *et al*., 2006). Maany sand beach injury assessments have also ignored the interstitial beach meiofaunal community, which possesses exceptionally high diversity at both phylum and species levels. Nevertheless, this group of small invertebrates living among the sand grains has not been shown to serve as important food sources for the higher trophic level consumers of value to the public. Biodiversity itself has value, however, and this meiofauna may hold unrecognized importance.

2.3 Quantifying losses by resource equivalency analysis (REA)

When scaling methods based on biological metrics are applied to organisms rather than habitat, the term "Resource Equivalency Analysis" or REA is often used. The sections below describe REA methods for injury to fish, birds, reptiles, and mammals.

2.3.1 Fish

Fish injuries from oil spills are typically assessed by using a modeling approach that combines fate of spilled oil and a biological effects submodel to compute fish injuries. The SIMAP program (Spill Impact Model Application Package) is a proprietary model that hindcasts physical fate of spilled oil and connects the oil exposures arising from that modeling with a biological effects model to estimate injuries to fish in the water column (French McCay, 2002, 2003; French McCay *et al*., 2004). The physical processes included in SIMAP are slick spreading, evaporation of volatile components from the surface, randomized dispersion, emulsification, entrainment of droplets into the water, dissolution of soluble compounds, volatilization from the water column, partitioning, sedimentation, stranding on shorelines, and degradation. The program models surface slick dispersion and dynamics, water column dissolved concentrations of aromatics over space and time, and sediment concentrations. For the water column, lower molecular weight aromatics are dissolved from whole oil and partitioned in the water column and sediments.

The biological effects submodel hindcasts fish mortality in the water column based on laboratory information on acute toxicity (LC50). The model estimates the area of the water surface, the volume of the water column, and the proportion of the fish population affected by oil components in water (as well as on the surface and in the sediments for assessing toxic mortality of other groups of organisms). Extent and duration of exposure are outputs for MAHs and PAHs, allowing computation of acute toxicity to each component. The losses are integrated over space and time, summing across toxic compounds, producing a number reflecting the total percentage of the fish population killed from acute exposure. This percentage is computed separately for fish eggs, larvae, young-of-year, juveniles, and adults. From information on abundances of fish populations in the spill area, then total acute mortalities of each life stage are computed. With published age-length relationships, these numbers of deaths are converted to fish biomass lost for each species or group of species. Then the biological effects submodel applies demographic fish population dynamics modeling to compute how much additional fish production is foregone by the untimely deaths preventing the fish killed in each life stage from completing their expected life span. This demographic model includes age-specific mortality and growth rates necessary to project expected growth and production of all those individuals that were killed by oil exposures.

Application of this model to determine injury is illustrated by the damage assessment for lobsters after the North Cape oil spill (French McCay *et al*., 2003a). This case is especially notable because the benthic habits of lobsters and low mobility allowed field sampling of lobsters in and out of the spill area to serve as an evaluation of the accuracy of the SIMAP modeling method. The model produced an estimate of 8.3 million dead lobsters, as compared to 9.0 million computed empirically from by field airlift sampling and visual quadrat counts. Consequently, this analysis provides a measure of comfort with the acute mortality modeling approach for assessing fish mortality. The proprietory nature of the SIMAP program, however, inhibits the ability of others besides the owner to make adjustments of parameters and to conduct formal sensitivity analyses, representing one limitation of this currently accepted approach. Because collection of dead organisms after a spill rarely includes all those that died, and fish tend to be mobile and difficult to observe and count, modeling seems more defensible as an approach to assessing water column mortality and total injury to fishes. However, some circumstances allow inclusion of empirical data taken in the wake of the spill. An acidic phosphate process water spill into the Alafia River, which flows into Tampa Bay, allowed collection of dead fish to provide the basis on which to estimate total biomass killed and production foregone. This Mulberry Phosphate case allowed the use of empirical data because the dead fish floated and were constrained within a relatively small basin.

2.3.2 Birds

Bird injuries are common after oil spills because many waterbirds use water surfaces for foraging and floating and shorelines for feeding and nesting: the water surface and shoreline are each at high risk of exposure to oil. In addition, birds, especially apex predators, are susceptible to reproductive and other injuries from ingestion of toxic chemicals in their prey. Like other vertebrates, birds are valued resources in natural ecosystems, so natural resource injuries to birds are frequently assessed after environmental incidents. Similarly, REA analyses are commonly applied to determine how to provide quantitative restoration for bird injuries.

Many NRD cases involve bird injuries, indicative of their high susceptibility to floating pollutants like oil, contaminated shorelines, and toxicity of their prey. Many assessments of injuries to birds include similar elements: (1) collection of contaminated dead birds by species (or genus if species identification is rendered unreliable by contamination or decomposition) in surveys of the affected area; (2) estimation of the proportion of dead birds that is not located and recovered during the survey; (3) capturing and counting of evidently contaminated birds, often for transport to decontamination centers; (4) estimation of survival probability after decontamination; (5) determination of any impacts on nesting and reproductive success; (6) tissue analysis to assess concentrations of contaminants that may induce injury; and (7) life history and population reviews to infer how quickly natural recovery would occur and whether nesting habitat, predators, or numbers of breeding birds limit population sizes.

Here we describe two bird injury assessments, chosen to illustrate methods applicable to two different, commonly encountered situations. First, we present injury assessment methods applied to piping plovers after the North Cape oil spill (Donlan *et al*., 2003) to illustrate a response to assessing injury to a species that is federally listed as threatened or endangered. Second, we describe injury assessment of a suite of seabirds, pond birds, and terrestrial birds also injured in the North Cape oil spill (Sperduto *et al*., 2003). This group of birds does not include any listed species, so the injury assessment and the matching restoration project are not subject to the same constraints as applies to a listed bird. Many other studies involve assessment of bird injuries, including waterbirds in the Julie N case, seabirds in Summer Bay, Unalaska Island in the Kuroshima case, and many others (Tampa Bay, Stuyvesant, Westchester, Luckenbach mystery spill, Commencement Bay, Florida mystery spill, and Chalk Point NRD cases).

2.3.2.1 Injury assessment to listed or highly valued bird species

Plover injury in the North Cape spill was assessed by evaluating how productivity of chicks changed from the year before the oil spill to the year after the spill (1996) at the oiled and at 4 unoiled reference beaches. Productivity of fledglings declined by 37% from 1995 to 1996 at the oiled beach while remaining unchanged on average at the reference beaches (Donlan *et al*., 2003). To confirm that this change was likely to be a spill effect, abundance of invertebrate prey and plover foraging behavior was assessed in summer 1996 after the oiling. Amphipod prey in the wrack, a taxon highly sensitive to oil toxicity, were substantially lower on the oiled beach than on reference beaches. Probably in response, the average daily distance traveled by foraging chicks to obtain food was much greater on the oiled beach in 1996 than in the two previous years. Consequently, the observed decline in plover productivity could be mechanistically linked to impacts of the oil on its prey and added costs and risks of chick foraging.

The quantitative injury was computed as the decline in chicks produced by the 9 nesting pairs on the oiled beach from 1995 to 1996, namely 5 chicks (Donlan *et al*., 2003). Applying the documented over-winter chick survival of 48%, 2.4 of those missing 5 chicks would be expected to survive to the next breeding season when at 1.56 chicks per pair they would be expected to produce 1.87 more plover chicks. This process could be continued indefinitely to include an infinite number of generations. If the numbers of breeding plovers, not habitat, food, or predators, were to limit plover population growth, then the total injury would grow over time in this fashion. However, such injuries to subsequent generations were not added to the initial year's loss of 5 chicks because in-kind restoration providing at least 5 additional chicks would likewise be expected to yield the same numbers of added progeny in subsequent generations, thereby balancing losses and gains after taking into account time lags by discounting (Donlan *et al*., 2003). This balance is achievable because in-kind compensatory restoration was applied. If restoration were achieved in other ways, this component of future injury would need to be added for each year that the number of breeding plovers is thought to limit plover population growth. Consequently, injury assessments like this one for piping plovers require challenging inferences about what factor limits population growth of the injured species and for how long that factor will continue to operate as the species population grows. When the injury involves a Threatened or Endangered Species, however, existing species recovery plans exist to help guide restoration planning.

2.3.2.2 Injury assessment to guilds of birds

Sperduto *et al*. (2003) describes the quantification of injury and compensatory restoration of all birds injured by the North Cape oil spill except the piping plover. That included seabirds, salt pond birds, and terrestrial birds. The basis for quantifying injury to these birds, as in many other cases, was to conduct a search for dead and dying birds, estimate the proportion of killed birds that remained undetected and unrecovered, and calculate the sum of bird production lost by direct mortality plus production foregone because of the additional loss of the progeny that would have been expected had these dead birds survived. The numbers of dead birds collected after an oil spill or analogous environmental incident is an underestimate of birds killed because of carcass sinking, transport away from the spill area searched, and scavenging. On rare occasion, experiments are done by releasing marked carcasses to estimate the loss rate. Physical transport models have also been used to estimate degree of carcass retention versus export (e.g., Ford *et al*., 1987, 1996). In the North Cape example, the method of estimating underdetection involved a review of the literature and informed choices of underdetection rates for ecologically similar birds under similar environmental circumstances (Sperduto *et al.*, 2003). In most oil spill incidents, oiled yet still living birds are captured and brought to rehabilitation centers for cleaning. An estimate of the (large) numbers of oiled birds that fail to survive during and after this rehabilitation attempt must be added to the numbers of dead birds collected (see the Chalk Point record in NOAA *et al.*, 2002 for an example) to seed the modeling process used to compute the total numbers of birds killed (those detected plus those undetected).

The estimated numbers of birds killed by the oil spill, including adjustment for those that were not detected, did not constitute the complete injury for the injured birds, except for the taxa where recovery was estimated to occur within one year. Instead, the interim loss of birds was computed in units of bird-years foregone. This metric involves first gathering information on the life history of each injured species or higher taxon (its production rate of fledglings, maximum life span, stage-specific natural survival rates, and current population abundance) and on the numbers of birds killed directly by the spill. From this information, Sperduto *et al*. (2003) divided the injured birds into two groups, those for which recovery in abundance was likely within one breeding season and those requiring longer time frames. For those taxa anticipated to recover within one breeding season, the estimated total number of birds killed was equivalent to the bird-years foregone. Species requiring longer time frames than one year for natural recovery (loons, eiders, grebes, scoters, goldeneyes, mergansers, and bufflehead) are typically characterized by longer times to reach breeding age and lower reproductive productivity. Sperduto *et al.* (2003) presumed that natural population recovery for species of birds in this group would be achieved in a period of time equal to the remaining expected natural lifetime of an average aged bird in the absence of the spill. Because of the inability to age carcasses of oiled birds upon recovery, the age of dead birds in this group of slow recoverers was estimated to be the average age in the population. This demographic parameter was computed on the basis of stage-specific (fledgling and adult) annual mortality rates and maximum life span, taken from multiple literature sources. From this computation of the average age of birds in each injured population and the annual mortality rates and maximum life span, the expected number of additional years each dead bird would have been expected to have lived was computed. For example, the average scoter was computed to be 3.8 years old, could live to a maximum of 11 years of age, has annual probability of survival as an adult of 0.75. From this life history information, a 3.8 year-old scoter would have been expected to survive for 2.87 more years. Total bird years lost by each species of bird killed directly by the spill was then computed in bird-years foregone by multiplying the numbers killed by the expected additional life span absent spill mortality.

For each species of bird whose recovery was judged to require longer than 1 year, the number of bird years lost through direct mortality was augmented by adding to it the loss of bird years that would have been expected from first-generation fledglings that were then not produced by the dead birds (Sperduto *et al*., 2003). This required determining how many of the dead birds of each species or taxon would have been expected to attempt reproduction. That first involves computing the expected age distribution of each injured species. Then, by knowing the age of first breeding, the numbers of breeding birds killed was readily computed. The number of bird years lost by unborn fledglings per breeder for each breeding season was then determined by multiplying the number of fledglings produced per breeding individual by the average life span of a fledgling. That computation was repeated for each breeding season that any of the directly killed birds was expected to live to reach. The computations done by Sperduto *et al*. (2003) assumed that the numbers of breeding seasons involved was the number of additional years that an average-aged bird was expected to survive. Thus the grand total of bird-years foregone by fledglings never born to the birds killed by the spill was computed by summing the products for each of those years of the numbers of surviving breeders times the annual survival times the expected number of bird-years lost by unborn fledglings per breeder. This computation could have been made more precise by using the complete demographically expected age distribution of killed birds to calculate the expected number of breeders alive in each successive breeding season and then project those age classes ahead annually until none of the original population of killed birds survived. This method would allow bird years lost by unborn fledglings to be based not on assuming that all birds killed were of average age but rather assuming a full distribution of age classes at death. However, this approach would require changing the presumption that recovery of these slow recoverers would be complete after the expected remaining natural lifetime of an average aged bird.

This method of computing the component of bird-years foregone by the absence of the first-generation fledglings that would have been produced by the birds that died in the absence of the spill can be further complicated by varying life history and reproductive behavior. One considerations is the existence in many birds of groups of non-breeding adults, which if breeders have been lost can then step up to breed. This response would mean that the losses of fledglings in the breeding season(s) following a spill would not be so large. Common loons fall into this category of birds: 20% of those killed by the North Cape spill were assumed to be non-breeding adults (Sperduto *et al*., 2003). Thus, up to 20% of the loss of breeding adults could enjoy partial reproductive compensation by these non-breeders filling in reproductively. However, the compensation is not complete because these inexperienced breeders have reduced success. In the case of loons in the North Cape injury assessment, the contributions of non-breeding adults to reproduction were assumed to be negligible for the first two years and then serve fully for reproductive seasons thereafter (Sperduto *et al.*, 2003).

2.3.3 Reptiles

As in other resource injuries, there are two generic types of injury assessment. One type treats injury by species and applies a demographic model to compute total injury in terms of numbers of individuals at a certain life stage that must be added to the population or protected from certain death to replace lost individuals and result in no net loss to the population. This approach is most applicable to injury involving an endangered, threatened, rare, or especially valued species. The other type treats a guild of species in the affected taxon, does not make species distinctions, but instead estimates the lost production as biomass of the dead plus production foregone after the dead failed to complete their natural life span and the additional growth they would have experienced. In both types, there is potential for the mortality event to have negative consequences to production of individuals or biomass in one or more subsequent generations, but this possibility is not always explored in injury assessments. For reptiles, it is difficult to conceive of an example where a guild of reptiles was so common that the guild approach would be adopted, although some freshwater habitats may contain numerous species of turtles that are not targets of conservation plans. In marine and estuarine environments, all sea turtles are threatened or endangered, and alligators and terrapins are also species of concern that would suggest a species-specific demographic approach to injury assessment.

In the Fort Lauderdale Mystery Spill (NOAA and FDEP, 2002), sea turtle injury was assessed by using the SIMAP (Spill Impact Model Analysis Package) routine described earlier for fish injury assessment. SIMAP is a physical fates model that is linked to a biological effects sub-model. In this application, oil was discovered washing ashore along the southeast Florida coast from North Miami Beach to Boca Raton in the first week of August 2000. At this time substantial numbers (estimated at 137,500) of hatchling sea turtles of at least three species (loggerheads, greens, and leatherbacks) had left the beach nests and were located in nearshore waters west of the Gulf Stream. Other adult (77, mostly female) and juvenile (327) sea turtles were also present in this region of the sea, plus many nests have not yet yielded their hatchlings by that date so they are at risk on the ocean beaches. The SIMAP process was initially run backwards to hindcast the location and spread of the spill, so as to infer the time course of the oil slick across the surface of the ocean. From that physical fate modeling, the number of sea turtle deaths was estimated from these estimates of sea turtle abundance in the spill's path at that time and the assumption that 50% of the hatchlings in the wake of the slick were killed by smothering and toxicity because of their small size and surface habits, whereas only 1% of adults and juveniles in the wake suffered mortality because they spend most of their time underwater and are of larger size. The modeling using these assumptions implied deaths of 7,800 hatchlings and less than 1 adult and 1 juvenile sea turtle. Using the demographic parameters of age-specific survival, the 1 adult or juvenile sea turtle lost was converted into hatchling equivalents, which was 357 hatchlings, making a total sea turtle mortality of 8,157 hatchling equivalents. The species composition of the dead was assumed to match the area's general distribution – 86% loggerhead, 14% green, and 0.1% leatherback.

The Chalk Point oil spill in Swanson Creek in April 2000 resulted in collection of 12 dead diamondback terrapins (Byrd *et al*., 2002). Injury assessment was conducted in three steps. First, a field assessment of nesting success during spring 2000 provided information on whether oiling of the shoreline where nesting occurs caused any reduced reproductive success in the first spring following the spill. Second, a population risk assessment model was applied to estimate the numbers of terrapins suffering from acute mortality. Third, unlike the sea turtle assessment for the Fort Lauderdale Mystery Spill, the trustees for Chalk Point included terrapin losses in the second generation resulting from the reductions in numbers of adult breeders in 2000.

The study of effects of oiling the shorelines on terrapin nesting success involved constructing 50-m^2 enclosures on heavily oiled and unoiled shores and monitoring the numbers of terrapin hatchlings that emerged by September 10, 2000. Then several enclosures on both heavily oiled and unoiled shores were excavated to identify nests from which hatchlings had emerged and to infer mortality. Clutch size was found to be lower in 2000 at both oiled and unoiled sites than in that same region in the 1978-1991 period before the oil spill. Researchers found a significantly higher frequency of dead embryos in nests on oiled shores and a lower percentage of spring hatchlings. The numbers translated into a 10% reduction in numbers of spring 2000 hatchlings, which demographic projections determined represented a loss of 836 (discounted) terrapin-years. The risk assessment estimation of acute mortality was based on density estimates of numbers of terrapins per km of shoreline (86), the lengths of heavily oiled, moderately oiled, and lightly oiled shoreline, and presumed mortality risks for each level of oiling (10%, 2%, and 0.5% respectively). This computation produced a total number of adult and juvenile deaths of 122, which demographic projections shows equivalent to loss of 616 terrapin-years. No attempt was made by use of recovery studies to confirm that only 12 of these would have been expected to be located by the post-spill surveys. A further modeling projection by application of demographic rates of reproduction and survival showed an additional loss of 3,793 (discounted) terrapin-years from the reduction of animals in the next generation. It is interesting to note how large this contribution to total terrapin injury is.

2.3.4 Mammals

Injury assessments for mammals are likely to be done as REAs based on demographic projections, as opposed to aggregating them into a guild and basing injury on lost production. Mammals are highly valued and the development of a species-specific injury assessment of how many individual losses the population has suffered by mortality and reproductive losses is likely to be justified. In addition, mammals have been the target of sufficient research, often including development of species recovery plans, that demographic information is readily available on which to construct the demographic model required for compelling species population analysis of injury.

As part of the injury assessment conducted in the salt marsh habitat after the Chalk Point spill, an assessment of spill impacts was conducted on muskrats (NOAA *et al*., 2002). Muskrats represent a species of mammalian consumer sustained by salt marshes in the Chesapeake Bay region, providing a target for trappers, and deserving of a species-specific REA. A total of 70 muskrats was discovered dead after the spill, placing a lower limit on total mortality. A literature review implied very high sensitivity of muskrats to oiling because oiled fur loses its thermal insulation capacity and removes buoyancy for this aquatic animal. Thus, the trustees assumed acute toxicity upon contact with oil. Muskrat densities were estimated for Swanson Creek marsh and the wetlands of two other less extensively oiled creeks. GIS mapping of muskrat habitat in each creek allowed multiplication of muskrat habitat times density to yield total estimated deaths of 376. Muskrats are highly fecund and appear to be at equilibrium population levels in these marshes, such that recovery of these 376 lost animals was anticipated within 2 years. Based on these conclusions, the REA approach was applied to compute the muskrat-years lost (373) from the spill and the acres of restored marsh habitat, assuming quality equivalent to Swanson Creek and supporting 4.2 muskrats per acre, required to compensate quantitatively for those lost muskrat-years.

2.4 Quantifying restoration benefits in HEA/REA models

Like injuries from environmental incidents, benefits of restoration options must also be quantified so as to achieve appropriate compensation for losses. Such computations of restoration benefits are simplest when they represent in-kind restoration and use the same units, such as acre-years of salt marsh, secondary production of marine invertebrates, numbers of individuals of a specific species, or bird-years for a group of birds. However, some injuries do not lend themselves to in-kind replacement. For example, restoration of injury to intertidal sand flat habitat is unlikely ever to be achieved by grading and flooding shoreline to create more tidal flats. Instead, a higher-value habitat will be restored or created in a fashion that presumably replaces the quantitative level of loss of sand-flat ecosystem services with similar but not identical ecosystem services of that other habitat. Similarly, replacing lost production of some guilds of animals such as riverine fishes may not be feasible so replacement may involve augmenting production of similar but different fishes in a high-value habitat like a salt marsh or oyster reef. Here we provide examples of alternative types of restoration, including the simple in-kind replacement of lost habitat or lost resource and the more complex out-of-kind restorations, showing how each method applies scaling to match the respective injury.

2.4.1 Marsh creation

Restoring tidal marshes represents an appealing method of compensating for the loss of a wide range of different habitats and the ecosystem services that they provide. Marsh restoration can even be scaled to replace resource losses to guilds like predatory fishes or to individual species populations dependent on marsh habitat. Tidal marshes have been dramatically depleted in the U. S. and worldwide, so their reconstruction makes sense from a perspective of restoring historical baselines and natural balance among interconnected estuarine habitats. The knowledge of where tidal marshes have existed in the past provides guidance for where marshes may prosper if restored. The methods of *Spartina* marsh restoration have been well worked out and there is a substantial literature on how to design and install a new salt marsh (*e.g*., Seneca *et al*., 1985; Broome *et al*., 1986). Tidal marshes are recognized as providers of numerous ecosystem services (MEA, 2005; Peterson *et al*., 2008a), so scaling by use of a proxy to replace one injured service can often be assumed to replace several. Finally, marshes are highly productive so that compensatory restoration can be done efficiently on relatively few acres, offering cost savings.

The most straightforward scaling of marsh restoration to a service loss involves in-kind replacement of lost acre-years of marsh ecosystem services (SAYs). Such restorations are extremely common, especially as mitigation for unavoidable marsh injuries during land development. Replacement of lost marsh by new marsh involves few assumptions because the same habitat is involved on both sides of the ledger. The greatest challenge comes in projecting how rapidly the full suite of marsh ecosystem services will return after restoration and what the maximum percent of ecosystem services will be at equilibrium. The need for this critical knowledge exists even when marsh restoration is conducted to restore some other resource, not marsh habitat itself. The most reliable means of ensuring true compensation would be to monitor the injured marsh so as to test assumptions about the trajectory of recovery of the injured marsh after any primary restoration is conducted, and to monitor the same metric(s) in the restored marsh as a test of assumptions about development rate of services in both the injured and newly created marsh habitat. Unfortunately, the responsible parties who must fund the compensatory restoration dislike uncertainty and wish to settle quickly on restoration plans and costs so that incorporating an expensive long-term monitoring of injured and restored marshes to ensure quantitative compensation is not generally feasible. Contingency funds as a percentage of total compensatory restoration costs are collected from settlements of injury liability cases that can provide funding for limited monitoring and limited mid-course corrections of restoration projects. However, conducting more in-depth monitoring in selected systems to allow improvement of estimates of the time course of recovery of ecosystem services in injured and restored marshes can be done and could improve the projections made in future applications of restoration scaling for tidal marshes.

Michel *et al*. (2008) have indeed monitored both injured and newly constructed salt marshes after the Chalk Point oil spill on Swanson Creek, Maryland (NOAA *et al*., 2002). Marsh restoration was completed in October 2005 and Michel *et al*. (2008) monitored the development of the *Spartina alterniflora* and *S. synosuroides* marsh in replicate locations 2 m from the creek margin in September 2007. Samples of stem density and height of above-ground vegetation and below-ground biomass density at two depths provided the metrics for assessing recovery of marsh ecosystem services, analogous to the injury metrics applied earlier. By comparing the magnitude of each metric in the restored marsh to the levels exhibited at that same time in natural unoiled marsh, Michel *et al*. (2008) produce quantitative evidence on which to evaluate the initial assumptions about development time of marsh services in the compensatory restoration project. *Spartina* stem heights increased in those two years following marsh creation to 95% of natural marsh. Stem densities were 108% of natural marsh values. On the other hand, below-ground biomass reached levels of only 39% of natural marsh at the 0-10 cm depth and 7% of natural marsh at 10-20 cm depth. These metric values compare to projections of services development in the restoration scaling for the restored marsh of 50% after 5 years, 75% after 10 years, and 80% after 20 years (NOAA *et al*., 2002). It seems unlikely that the below-ground services of biogeochemical cycling will achieve these projected levels, whereas above-ground services may already have met the anticipated targets well before anticipated dates. Michel *et al*. (2008) recognize that the soils at the restoration site were very inorganic and impenetrable. Careful choice or preparation of more organic-rich soils may thus be important for future *Spartina* restorations to achieve assumed rates of return of all ecosystem services.

Salt marsh restoration has also been used in compensatory scaling for other types of lost services, both habitat injuries like oiling of intertidal or subtidal flats and injuries to resource guilds or species of animals. For both types of compensation the scaling is more complex than in-kind scaling of lost marsh to restored marsh. The use of salt marsh restoration to compensate for loss of ecosystem services of subtidal bay bottom in the Lavaca Bay Trustee Council (2000) case record illustrates the scaling that converts one habitat to another, in this instance based on informed opinions of experts, who concluded that an acre of fully functional brackish salt marsh provided 5 times the ecosystem services as an acre of bay bottom. Consequently, the lost SAYs of bay bottom were replaced by fully functional acres of restored salt marsh in a 5 to 1 ratio. Newly available habitat conversion ratios based on quantitative productivity ratios computed across all three lowest trophic levels (Peterson *et al*., 2009) represent a more systematic method of scaling future habitat conversions that lead to salt marsh restoration. These conversion ratios taken from synthesis of production data across three trophic levels suggest that *Spartina* marsh provides about 1.7 times the per acre services as intertidal flat habitat and 2.1 times the services of shallow subtidal habitat.

Scaling of injuries to individual species of injured animals by using salt marsh restoration can be done in a variety of ways. One is well illustrated by the scaling methodology used to compute how marsh creation would compensate for muskrat injuries from the Chalk Point oil spill. We describe the details of this procedure in the section on mammal injury. The first step of this REA was to conduct the injury assessment on the muskrat population, computing how many muskrat-years were lost through acute mortality from oil exposure and from muskrat losses in subsequent generations because of a smaller parental population. The numbers of muskrat-years lost was then matched in scaling marsh restoration by using local information on average muskrat density per acre of *Spartina* marsh and assuming that muskrats would attain that same density in the restored marsh. The marsh is used by the muskrats for nesting and provides food resources. After accounting for discounting over time, this computation serves to identify the acreage of restored *Spartina* marsh required to yield the lost muskrat-years.

A final type of scaling compensatory restoration of marsh to animal injuries addresses injury to an entire guild of animals. Thus instead of applying some form of demographic analysis to compute injury at a population level, injury is computed in the form of production lost and foregone by the guild of species, against which compensation is calculated by expected augmentation of production. The Lake Barre, Louisiana oil spill case (Penn and Tomasi, 2002) provides an example of this approach. Injuries to aquatic animals (fish, squid, shrimp, crabs, and other invertebrates) were computed by applying a model to estimate acute mortality from exposure to lower molecular weight PAHs and adding to the biomass of the animals killed the biomass of production foregone by those dead animals that failed to complete their natural life spans, thereby representing the total lost production. Bird production losses were also estimated by modeling all bird encounters with the floating oil, assuming acute mortality of all birds encountering the floating oil, and computing total production loss by assuming that each dead bird weighed 1 kg. The computation of bird production loss includes no component for production foregone by untimely mortality, justifiable because these birds rapidly achieve adult size and grow no more through later life. The production losses in each of these guilds of animals was compensated using a HEA to compute the acreage of marsh restoration required to produce incremental production of aquatic animal and bird biomass to equal the losses. This scaling was based on *Spartina* production at the primary producer level and then applying ecological energetic efficiencies to convert that primary production of vascular marsh plants first to detritivores at 4% efficiency, then to fish and invertebrate primary predators at 20% efficiency, and then finally to bird secondary predators at 2% efficiency. This application of energetics thus allows computation of how many acres of fully functional *Spartina* marsh are required to match the observed production losses at higher trophic levels, accounting for the trophic level on which the lost production occurred.

This process of using salt marsh restoration to flow upwards trophically to restore lost animal production in the marsh habitat is a common practice. The specific application of the concept of scaling marsh restoration to replace lost animal production in the Lake Barre case deserves some further comment. First, the losses to each guild were combined additively, with separate marsh acreage required to replace production losses of each guild. This follows the practice explained and defended in French McCay and Rowe (2003). Second, the scaling incorporates interesting complexity that has generic applicability. First, instead of restoring *Spartina* by planting the entire dredged platform on East Timbalier Island, only strips were planted, with the expectation that the *Spartina* would spread into the unplanted areas. This practice reduces the cost of transplanting but at the cost of stretching out the time table of achieving maximal coverage and service provision on the unplanted areas. Planted areas are assumed to take 5 years for development of maximum marsh plant services (which are assumed to equal only 50% of natural marshes), whereas the unplanted areas are assumed to develop to their maximum only after 12 years. In each case the marsh is projected to have a 25-year life span. Because *Spartina* marsh would also be expected to develop on the dredged platform even in the absence of overt restoration, achieving completion after 20 years, only the difference in service gains from restoration can be credited as compensation for injuries. In addition, the installation of the strip closest to and parallel with the shoreline is also assumed to slow the erosion rate by stabilizing the sediments and elevating the soil level. Consequently, the restoration scaling includes gradual erosion of unrestored platform and its more slowly developing marsh and faster rates of erosion of this unrestored platform.

This erosion component of the scaling computation was explicitly parameterized for the Louisiana island on which the restoration was to be conducted. The coast of Louisiana is subsiding, so consideration of erosion of marshes in scaling restoration benefits is a necessity there because time frames over which the marsh might reasonably persist are long enough that erosion will realistically influence the functional area of marsh that does persist. One might assume no need for such considerations in other geographic regions without the high subsidence and without engineered deflection of the Mississippi mainstem that diverts sediment that could help elevate the sinking marsh. However, growing rates of sea level rise now make shoreline erosion a concern in many other geographic regions, especially south Florida and North Carolina, so erosion consideration in marsh restoration scaling is broadly appropriate using locally appropriate rates of relative sea level rise. Furthermore, marsh plants elevate the bottom by trapping inorganic sediments and by below-ground production of roots and rhizomes (Morris *et al*., 2002). Consequently, modeling of marsh erosion over the time frames required for computing benefits of restoration scaling should best address the balance between relative sea level rise and plant-induced elevation of the soil surface. In addition, salt marsh naturally transgresses up-slope and landwards as sea level rises. Marsh restoration projects that are created seaward of structural barriers like bulkheads are blocked from up-slope transgression and over time frames in excess of about 20 years, based upon present increasing rates of sea-level rise, will lose marsh area and thus provide annually decreasing ecosystem services (Peterson *et al*., 2008b).

The Lake Barre restoration scaling of salt marsh to replace lost services from guilds of marsh animals is based only on the vascular plant production of the *Spartina* grass. The vascular plant production forms only one of two bases of food chains originating in the salt marsh. Benthic microalage are also important contributors to salt marsh food chains and they can be consumed directly by herbivorous invertebrates without passing through a microbial intermediate of fungi or bacteria. The consequence of this more direct transfer of energy to the second trophic level is a higher ecological efficiency of transfer (Kneib, 2003). French McCay and Rowe (2003) provide an example from the North Cape oil spill of how this trophic support method of scaling translates primary production of both vascular plant and benthic microalgal production from salt marsh (and seagrass) habitat restorations to the higher trophic levels. In their transfer efficiencies, benthic invertebrate detritivore production is only 2.1% of vascular plant (*Spartina* spp.) production, whereas the benthic invertebrates grazing as herbivores directly on benthic microalgae achieve 10% of the production of the microalgae. This second pathway accounts for a large fraction of the total production of animals supported by primary production of the salt marsh. It is important to note, however, that benthic microalagal production in salt marshes and on intertidal flats is approximately equivalent (Sullivan and Currin, 2000), so the only net benefits in primary production and support of higher trophic levels from restoring a salt marsh on an intertidal flat would come from the addition of the vascular plant pathway. However, salt marsh restorations that grade down upland shores to create salt marshes would not initially possess benthic microalgae feeding marine food chains, so the initial status of the land on which the marsh restoration is achieved plays a substantial role in computing the net benefits in scaling the compensatory restoration (French McCay and Rowe, 2003). Because elevating intertidal flats to provide the geomorphology appropriate for establishment of vascular marsh plants would involve a dredge-and-fill project on existing valued habitat, the grading down of uplands is by far the more common practice.

The specific ecological efficiencies of transfer between consumer trophic levels in the North Cape application described in French McCay and Rowe (2003) differ from the classic Lindemann (1942) efficiencies of 10% by using more recent empirical data that reflect differences among various types of consumers. This information on transfer efficiency is continuously expanding, so review of the most recent evidence is appropriate for new applications of this trophic support scaling technique. French McCay and Rowe (2003) assumed a 10% ecological efficiency of conversion of benthic microalgal production into benthic herbivore production, but they also made an argument that could support a 20% level. Peterson *et al*. (2009) in their synthesis of production data at the lowest three trophic levels found empirical evidence for an efficiency of 26% from benthic microalgae and from the fungi and bacteria on vascular plant detritus to benthic macroinvertebrates. Thus a figure higher than 10% seems justified in such scalings. French McCay and Rowe (2003) then apply a 20% ecological efficiency from benthic invertebrates to nektonic fish and invertebrates at the primary predator trophic level. This does not differ greatly from Peterson *et al*.'s (2009) 27% empirical average. The ecological efficiency of energy transfer from the primary consumers to birds and mammals at the next trophic level is widely acknowledged to be low (1-5%), in part because of high respiration costs of homeotherms (e.g., Pimm 1982). French McCay and Rowe (2003) chose to use a 2% ecological efficiency for these links in scaling marsh restoration to a suite of bird injuries. The food web relationships between prey species and predators can also be used to form the conceptual basis for economic valuations of prey species that provide an energetic support service for higher-order predators. Based on the idea of Goulder and Kennedy (1997), Allen and Loomis (2006) developed a method for distributing the known willingness-to-pay valuation for a high-order predator to assign indirectly partial willingness to pay estimates for each of the prey it relies upon in proportion to the energetic contribution in the diet.

The scaling computations for restored salt marsh habitat in French McCay and Rowe (2003) differ from those described in Penn and Tomasi (2002) in parameters relating to the return of full functionality of the created marsh. Unlike Penn and Tomasi's (2002) assumption for the Lake Barre restoration that created marshes only achieved 50% of the trophic support function of natural marshes, French McCay and Rowe (2003) assumed that 99% of full production services would ultimately be achieved. Furthermore, French McCay and Rowe (2003) assumed that created salt marsh required 15 years to reach the 99% functionality level and modeled the shape of the functional development with a sigmoid curve on the basis that gains would be slow at first, speed up and then slow down again as an asymptote of full trophic support was approached. Essentially all other scaling of functional recovery of created salt marsh applies a linear recovery rate for ease of computation. The quantitative scaling estimate is not overly sensitive to this choice of shape of the recovery curve. However, the assumption of duration of the project's functional life does affect the scaling if the project does not persist beyond about 10 years (French McCay and Rowe, 2003). French McCay and Rowe (2003) assume a 100-year life span for the created marshes created to compensate for suites of bird, fish, and invertebrate losses from the North Cape oil spill. Relative to the effects on computing the time-discounted benefits of marsh creation, anything beyond about 100 years represents virtual perpetuity (Peterson *et al*., 2003b). The grounds for assuming such a long life time for the North Cape compensation project reflected the institutional commitment to monitor and manage the restorations to assure functionality indefinitely. Such commitments can have substantial impacts on reducing the total area of restoration required to match the scale of injuries. On the other hand, as sea level rises even more rapidly, salt marsh is at serious risk of erosion, so only those restorations in which the upland habitat is preserved as natural and available for up-slope transgression of the marsh can be assumed to last more than about 20-40 years, depending on local rates of relative sea level rise (Peterson *et al*., 2008b).

One important caution to applying existing metrics to scale the production value of salt marsh creation relates to the vascular plant species involved in the new marsh. Essentially all of the data available on trophic support of vascular plants growing in salt marshes comes from study of one genus, the *Spartina* grasses. Many other vascular plants contribute meaningfully to salt marsh flora. Most prominently, these include *Juncus roemarianus* along the Gulf and south Atlantic coasts, many low-relief succulents like *Salicornia* and *Sarcocornia* on the Pacific coast, and invasive species like *Phragmites communis* and others. As long as the restored marsh consists exclusively of native *Spartina* (perhaps even just *Spartina alterniflora*, which dominates most scientific study), then the available evidence on primary production and trophic use and transfer efficiencies apply. In cases involving these other vascular plants, use of *Spartina alterniflora* information produces great uncertainty in actual benefits provided. Use of local species-specific information would make for more confident scaling of the marsh creation.

HEA scaling to date has been based on enhancing ecosystem services within separate habitats. By restoring a particular habitat in close proximity to one or more other kinds of habitats restoration can increase the ecosystem services provided by that restoration because of connectivity favoring ecosystem processes that add value. Several restoration plans developed by trustees recognize these landscape-scale benefits of pairing habitats. In some cases the spatial conjunction of habitats is recognized quantitatively by affecting the longevity of a neighboring restored or pre-existing patch of habitat. For example, a fringing oyster reef was established in the Lavaca Bay Trustee Council (2000) compensatory restoration program to replace a reef lost to dredging for removal of contamination. By placing this oyster reef adjacent to the restored low marsh, the estimated lifetime of the marsh restoration could be and was enhanced to 50 years on the basis of the oyster reef services as a natural breakwater, protecting against erosion and loss of marsh habitat area. In addition to realizing benefits of oyster reef habitat protecting the salt marsh, the Lavaca Bay marsh restoration also contemplated channels that seamlessly connected existing mature marsh with the restored marsh, probably speeding recovery of many marsh functions in the restoration. The restoration plan for the Mulberry Phosphate acid spill injuries in the Alafia River flowing into Tampa Bay also involved pairing of salt marsh and oyster reef restoration projects (French, 1999). This combination has value in the form of enhanced connectivity for nektonic predators that can move between habitats as conditions change and thereby enhance growth and survivorship, but the scaling computations did not reflect this synergy. Micheli and Peterson (1999) demonstrated that inclusion of structural corridors of seagrass habitat between otherwise isolated oyster reef habitat and intertidal salt marsh facilitated foraging by blue crabs thereby enhancing crab production but reducing hard clam production because of elevated blue crab predation. This is but one of several estuarine studies of how landscape relationships among habitats, including restored habitats, help determine the ecological functioning (e.g., Hovel, 2003; Grabowski *et al*., 2005). Additional progress on the scaling of habitat processes in the presence of other neighboring habitats would improve the ability to incorporate synergies in future restoration plans.

2.4.2 Oyster reef creation

Oyster reef habitat provides a wide range of ecosystem services (Lenihan and Peterson, 1998; Coen *et al*., 2007, Grabowski and Peterson, 2007), yet this habitat has experienced dramatic declines over the past century (e.g., Rothschild *et al*., 1994). The causes of oyster reef loss include habitat damage from fishing gear, overharvesting of spawning stocks, mortality from disease, and degraded water quality (Lenihan and Peterson, 1998, Jackson *et al*., 2001, Lotze *et al*., 2006). Because of the high value per-acre of the ecosystem services of oyster reefs (Peterson *et al*., 2009) and the magnitude of historical losses of this habitat, where appropriate, oyster reef creation is among the most commonly considered compensatory restoration actions. The US Army Corps of Engineers and many states have established oyster reef restoration programs unrelated to mitigation or compensatory restoration, indicative of widespread recognition of the importance of bringing back this habitat and its flow of ecosystem services.

Oyster reef creation has been incorporated into many compensatory restoration programs, with scaling based upon many different rationales, all related to replacing lost production. However, the production on which scaling is computed varies from the entire oyster reef ecosystem, to specific trophic or taxonomic levels, to functional groups of taxa, to explicit combinations of injured species. Most use of oyster reefs for compensatory restoration involves conversions of injuries from a habitat other than oyster reef to provision of analogous resource or habitat services from restored oyster reefs. There are no good model case studies illustrating the simplest restoration scaling in which oyster reef is restored to provide replacement for destroyed oyster reef habitat. Such HEA replacements restore in kind, so except for incorporating discounting to address time lags between injury and restoration and determining the temporal trajectory of development of the full suite of ecosystem services after reef construction, the scaling is a simple one-to-one replacement of acres of oyster reef restored for acres lost. Other oyster reef restorations are scaled to provide compensation for lost ecosystem services of other habitats or explicit suites of species or trophic levels injured in other habitats. These restorations require more complex HEA or REA scaling computations.

Model cases exhibiting different aspects of scaling include compensation for injuries following the Mulberry Phosphate acid spill in the Alafia River (French, 1999), an oil spill from Chalk Point, Maryland into a Chesapeake Bay tributary (French McCay *et al*., 2002), the Athos/Delaware River oil spill, and a Tampa Bay oil spill (FDEP *et al.*, 1997). In addition, oyster reef creation formed a major contribution to the mitigation for expansion of the Craney Island port on the Elizabeth River as it enters Chesapeake Bay (Peterson, 2003). These examples of compensatory uses of oyster reef creation thus can be sorted first by whether the scaling of required reef size was based on need for quantitative compensation for interim ecosystem service losses between the date of occurrence of a discrete pollution or physical incident and the date when primary restoration (including natural recovery) has restored the injured resource to full functionality or whether the reef creation was scaled to provide mitigation for a pre-planned, usually permitted, project that permanently removed the ecosystem services of some habitat or natural resource. The difference between these applications involves only the time scales over which the injury persists and thus the time scales over which the restoration project must continue to provide the lost services. A second distinction among oyster reef creation projects intended to provide compensation for lost ecosystem services relates to which specific service lost formed the basis for scaling quantitative restoration. The oyster reef creation following the Chalk Point oil spill was scaled on the basis of lost secondary production of benthic macro-invertebrates, fish and shellfish, and birds (excluding one, the ruddy duck) (French McCay *et al*., 2002). Scaling of the oyster reefs designed as part of the restoration planned after the Mulbery Phosphate acid spill was based upon lost production of fishes and large mobile crustaceans (Peterson *et al*., 2003b). In the Craney Island case, the size of the created reef was scaled to match the quantitative losses in secondary production of benthic macro-invertebrates and meiofauna of the sedimentary bottom habitat plus macro- and micro-zooplankton of the water column habitat (Peterson, 2003). In each case, the reef creation is recognized as providing a wide suite of ecosystem services not solely the one(s) on which scaling is based. Yet, scaling on the basis of one major type of injury provides a method of ensuring that this major component of the total ecosystem injury is quantitatively matched by the corresponding service from the restoration project.

2.4.2.1 Use of hatchery-produced seed oysters to restore lost production of numerous aquatic resources

The injury assessment program for the Chalk Point oil spill (French McCay *et al*., 2002) included June and September sampling of the subtidal benthic invertebrate communities along the axis of the Creek into which the spilled oil flowed, in other nearby unoiled control creeks, and along the mainstem of the Patuxent River. When combined with historical benthic monitoring data from the Patuxent River, this sampling was extensive and intensive enough to provide reliable data on which to estimate the spatial scale of injury to the subtidal sedimentary macrobenthos. In addition, knowledge of the life histories of the taxa affected by the spill allowed inference on the likely duration of injury necessary to compute the estimated interim losses. Benthic samples were sorted by species and dry mass was measured for each of them. The availability of such biomass information facilitates estimation of production, the preferred parameter on which to scale losses, particularly at lower trophic levels that are valued highly for their service of feeding animals on higher trophic levels.

Biomass differences between the oiled creek and the control creeks in the June sampling for taxa that exhibited differences reflected the direct loss of production from oil exposure. Any remaining biomass differences observed in September were used to reflect the duration of injury from persistent toxicity. Bivalve mollusks suffered substantial losses, as did one small crustacean. The crustacean did not demonstrate recovery by September, reflective of the high sensitivity of crustaceans to toxics. Polychaete worms demonstrated increased biomass in the oiled creek, probably reflecting consequences of microbes utilizing the petroleum hydrocarbons as a food source, indirectly enhancing food supplies for some pollution-resistant polychaetes. Based on the annual nature of reproduction in these large estuarine bivalves and the empirical suggestion of recovery within a year, the biomass difference in June was considered a measure of the complete interim loss in their secondary production. In contrast, the crustacean is known to produce multiple generations in a single growing season and its failure to recover by the September sampling date led to the conclusion that multiple cohorts experienced injury, so the biomass difference observed in June was multiplied by the number of cohorts assumed from the literature on the life history of this species to be similarly injured. Finally, the higher biomass of the polychaetes was considered to represent a partial mitigation for the losses in the other two taxa. The increased production of polychaetes was not credited against the losses in bivalves and the crustacean on a one-to-one basis because the bivalves provide important water filtration functions and the crustacean a critical fish forage source. Thus only 50% of the enhanced production of polychaetes was credited. This represents an example of using more than just production as a consideration in valuation of net injury, where both gains and losses occurred from the oiling. The sum of the losses and modified gains in secondary production of subtidal benthic macrofaunal invertebrates thus provided quantification of injury to benthic resources, against which scaling of macrofaunal production of oyster reefs was used to provide compensation.

The losses of benthic macrofaunal production from the oil spill were concentrated in two taxa, bivalve mollusks and a crustacean. Accordingly, the scaling of oyster reef habitat to compensate for the benthic invertebrate losses was done on the basis of enhanced production of these two taxa, oysters (a bivalve mollusk) and a group of oyster-reef associated crustaceans (grass shrimp, mud crabs, amphipods, isopods and tanaids). Because the created oyster reefs were to be placed near the location of the injury, in lower salinity regions of a Maryland river system where oysters have declined dramatically in the past decades, this oyster reef creation project could not rely on natural oyster recruitment. Instead, the benefits of added oyster production were computed on the assumption of expected growth, survival, and therefore production of biomass of introduced hatchery-raised seed oysters (here at a density of 500 m^{-2}). Available data from nearby oyster restoration studies showed that these seed oysters could not be expected to survive beyond 5 years, and the compensatory scaling assumed a second seeding at the same density after 5 years. Thus all oyster production benefits and indirect benefits of oyster reef habitat provision on crustacean production were derived from the fate of these two seedings and declined to zero after 10 years. The computation of expected enhancement in production of the suite of crustaceans associated with oyster reefs was based upon biomass information available from those same studies of oyster reef restorations in this region of Maryland. Because the created oyster reef covers up a pre-existing benthic soft-bottom habitat, the quantification of restoration benefits must account for possible lost production of the original habitat to compute net production benefits. In this Chalk Point case, no subtraction was made for lost production of bivalves expected in the initial soft-bottom habitat because the shell matrix installed during oyster reef construction promotes higher densities and production of infaunal bivalves along the oyster reef margins than exist in unstructured sediments.

The oyster reef creation was also used to compensate for losses to fish and larger mobile crustaceans (blue and other crabs) and for losses to birds (excluding only ruddy ducks, for which a species-specific REA was deemed necessary because this avian species was judged likely to suffer multi-generational losses and deserve individual attention in restoration planning). The restoration scaling for fish and these larger predatory shellfish and for the suite of birds was based upon the trophic level at which the injured guilds (taxa sharing similar trophic position) feed and the literature guidance on what the ecological conversion efficiency is as benthic invertebrate prey is converted to fish, crab, and bird biomass at higher trophic levels. Based on published energetic conversion efficiencies, the fish and predatory shellfish (crabs) were assumed to consume the benthic production directly and produce biomass at a 20% efficiency. That is for every kilogram of fish production lost, it takes 5 kilograms of secondary production by the oyster reef bivalves and small crustaceans. The estimated fish and shellfish injury was produced by applying the SIMAP model of physical fate of the oil and biological fate of exposed fish and shellfish to compute the acute toxic loss of biomass and the expected loss of future production by those animals had they survived until they died of natural causes (French McCay, 2002, 2003). For birds, the credit from oyster reef creation was computed similarly, but using different conversion efficiencies and including two different feeding guilds of birds, benthic invertebrate consumers and fish eaters. For birds that consume the invertebrates, a 4% conversion efficiency from oyster reef bivalves and small crustaceans was assumed based on literature information. For fish-eating birds (a higher trophic level), this conversion efficiency was assumed to be 0.4%. The biomass of birds lost was computed by multiplying the estimated numbers of birds lost times the average weight of the bird and summed across all injured species.

2.4.2.2 Restoring natural oyster reefs to provide permanent mitigation for loss of secondary production from filling of subtidal sedimentary bottom and water column habitats

Like the Chalk Point scaling, the oyster reef creation for Craney Island (Peterson, 2003) was scaled on the basis of quantifying the gains in benthic invertebrate production to replace lost animal production in a habitat (oyster reef) different from that where injury occurred. However, several fundamental differences exist between the two scaling exercises, justifying detailed presentation of the approach used in both cases. The first distinction is that the Craney Island scaling must provide indefinitely long ecosystem services because the permanent removal of benthic and water column habitat by fill to expand a port terminal makes the oyster reef creation a mitigation project not a compensation for interim losses until recovery has occurred. Second, the location of this oyster reef restoration project for Craney Island mitigation provides some confidence in adequate natural recruitment of oysters onto shell cultch so that repeated seeding by hatchery-raised oysters is not necessary. This affects the method of computing production benefits of reef creation. Third, the Craney Island oyster reef creation was scaled to account for losses not only of benthic invertebrates associated with subtidal soft-bottom habitat lost but also of largely herbivorous zooplankton associated with the water-column habitat that would disappear. Fourth, the Craney Island computations of both injury and compensation included not only macro-invertebrates but also smaller invertebrates, the meiofauna and micro-zooplankton.

Because oysters recruit naturally to shell substrata where the Craney Island oyster reef creation was contemplated, the demographic approach of following the fate of a cohort of seed oysters is not required for such situations. Instead, the method used to estimate oyster production used a review of the oyster biomass (B) achieved on nearby created oyster reefs and then applied a relevant Production to Biomass (P/B) ratio from the ecological literature on oysters. Uncertainty over the successful persistence of a constructed oyster reef in the Chesapeake Bay, where declines in American oyster spawning stock biomass leads to doubts about future recruitment of oysters, was incorporated in Peterson (2003) by a conservative halving of the biomass (and thus productivity) of oysters achieved on average by successful reefs in the lower Chesapeake Bay. Because both the benthic invertebrate and water-column zooplankton production estimates incorporated the smaller invertebrates (meiofauna and microzooplankton, respectively) the Craney Island benefits scaling for restored oyster reefs also added components of production associated with all other oyster reef-associated invertebrates, namely polychaetes and other macroinvertebrates and meiofauna. Literature information was available providing empirical measurements of the proportion of total macrofaunal production on an oyster reef that can be attributed to oysters (81%) versus all other macroinvertebrates (Bahr and Lanier, 1981). Much greater uncertainty exists, however, over the meiofaunal production associated with oyster reefs. No empirical measurements appear in the literature. Consequently, Peterson (2003) was compelled to use very general relationships between macro- and meio-faunal production on marine hard- and soft-bottom habitats to compute the expected meiofaunal contribution. The typical failure to measure injuries to meiofauna and microzooplankton and the lack of empirical data on the meiofaunal biomass associated with oyster reefs, natural or created, raises doubts about whether future scaling should repeat the inclusion of these smaller invertebrates in oyster reef scaling.

2.4.2.3 Restoring oyster reefs to replace injuries to an entire guild of riverine-estuarine fish and nektonic invertebrates

The release of acidic process water by Mulberry Phosphate into the Alafia River, which flows into Tampa Bay, caused a substantial fish kill. The numbers of dead fish and large mobile nektonic invertebrates like pink shrimp and blue crabs were determined by field sampling that recorded deaths by species and size class within species. These observations permitted injury to be quantified as biomass killed for each fish and mobile invertebrate plus the production of biomass foregone by terminating the lives of each individual killed before it would otherwise have been expected to die. This loss of production was then scaled to benefits of creating oyster reef habitat. Because this scaling of benefits of oyster reefs was computed to replace lost fish and mobile invertebrate production, the scaling computation method did not use the same approach as had been developed for Chalk Point or Craney Island. The Chalk Point scaling was done on the basis of estimating total invertebrate production on an oyster reef, and then that result was projected up the food chain to fish by assuming that the fish fed on the oyster reef invertebrates and converted invertebrate production to fish production at a 20% ecological efficiency. The Craney Island restoration was scaled to total invertebrate (secondary) production rather than to higher trophic levels. For the Mulberry Phosphate acid spill case, scaling was done in a taxon-specific fashion explicitly and directly for fish and mobile nektonic invertebrates (Peterson, 2000; Peterson *et al*., 2003b).

The species-specific scaling of how fish production is enhanced by creating oyster reefs began with a comprehensive synthesis of all published and unpublished quantitative sampling data comparing densities of fish and mobile nektonic invertebrates by species on oyster reefs and on nearby unstructured bottom (Peterson *et al*., 2003b). Any species that failed to exhibit a majority of studies showing density enhancement on oyster reefs was excluded from further consideration and made no contribution to the estimate of enhancement of fish production by creating oyster reef habitat. The remaining species demonstrating quantitative density enhancement in a majority of studies were separated into two categories: those whose recruitment is enhanced by the reef (RE) versus those whose growth is enhanced by reef presence (GE). The RE species were determined by examining the 0-year-class recruits: any species with augmented density of reefs that recruited exclusively to reef habitat or whose life history showed functional dependence to hard reef substrata during its early life was considered recruitment limited. In those cases, the complete expected lifetime production of the fish recruiting to the created oyster reef was credited to the reef habitat, even if, like gag groupers, the fish left the reef after an early nursery phase and completed life elsewhere. The other species whose densities exhibited enhancement on oyster reef habitat were considered to gain benefit from reefs by the reef structure allowing them protection to feed on reef-derived benthic foods. For these species, an index of reef exclusivity (IRE) was computed on the basis of published gut contents that ranged from 1.0 if all foods were reef-associated to 0.10 for planktivorous species that used the reef for shelter from which they fed in the water column. This index was then later applied to each species of fish and nektonic crustacean so as to develop an estimate of production credit that conservatively included only the fraction of expected lifetime growth and production directly attributable to the oyster reef.

Once the full set of RE and GE species was identified, then the density enhancement of each was used in combination with age-specific growth, age-specific mortality, and length-weight relationships from NOAA NMFS species profiles, FISHBASE, other standard data bases, and published papers to compute their expected lifetime production. For the GE species the reef credit was depreciated to the degree that the IRE fell below unity. By summing over all fish and mobile crustaceans, this computation yielded the expected augmentation in fish production attributable to an acre of oyster reef. Because the created oyster reef covers up a pre-existing habitat, typically unstructured and unvegetated intertidal bottom, the complete credit for creation of an acre of oyster reef habitat must involve subtraction of the fish production associated with that now replaced habitat. This was achieved automatically by the method of recording density augmentation on reefs as compared to unvegetated sedimentary bottoms and using that augmentation as the fish production benefit provided by reef restoration. The computations (Peterson, 2000; Peterson *et al*., 2003b) applied broadly by species (taxon) to the entire mid-Atlantic, southeast Atlantic, and Gulf coasts, and provided species-specific estimation of augmentation. Consequently, this analysis is an important one that can be applied broadly to future restoration scaling needs. This method is exceptionally dependent on the data synthesis and requires substantial effort in complex demographic computations for each species. Consequently, it can be justified only if the fish are sufficiently important to deserve special attention and unusually high levels of confidence in their restoration.

2.4.3 Mangrove creation

Mangrove forests provide ecological, social, and economic benefits of substantial value. Mangroves provide habitat for a large number of mollusks, crustaceans, birds, insects, reptiles, and even mammals, as well as generating subsistence and currency income from the harvest of some of these resources (McLeod and Salm, 2006). In addition to providing an important nursery habitat to sustain fisheries and support other exploited resources like mangrove crabs, the mangrove trees are also harvested for fuel, charcoal, timber, and wood chips (McLeod and Salm, 2006). Intact mangroves also provide an important protective barrier to habitats and human communities that reduces damage, injury, and death from tropical storms and tsunamis. The annual economic value of a mangrove forest has been estimated to be $200,000 - $900,000 US per hectare, based only on the prices of products and services that they provide (Wells *et al*., 2006).

Only meager information exists to compare the ecosystem functioning of restored mangrove habitat to natural or even partially modified mangrove habitat (Kaly and Jones, 1998). There are no good estimates of the time course of developing ecosystem services after restoration of mangrove habitat (Hawkins *et al*., 2002), so scaling mangrove restoration to compensate for lost mangrove forest services is highly uncertain (Zedler, 1999; Zedler and Calloway, 2000). The case of the 1993 Tampa Bay oil spill offers some guidance on in-kind compensatory restoration scaling of mangrove forest habitat, while the Fort Lauderdale mystery spill (NOAA and FDEP, 2002) provides more detailed methodology to support its scaling of mangrove habitat restoration to replace lost production of fish and invertebrates.

2.4.3.1 In-kind restoration to compensate for mangrove habitat injury

Trustees of the Tampa Bay oil spill case elected (FDEP *et al.*, 1997) to compensate for the loss of mangrove forest with in-kind replacement of injured mangrove habitat within by expanding mangrove habitat within the local ecosystem (Boca Ciega Bay). This option was considered most effective in matching those ecosystem services that were lost due to the oil spill. The restoration site was to be graded so that hydroperiod (tidal exchange) and elevation would allow planting of salt marsh grasses (*Spartina alterniflora*) and both the natural recruitment and supplemental planting of mangrove trees. This method facilitates development of mangrove forest by establishing a transitional earlier succession salt marsh: this worked well in previous mangrove forest restoration efforts. While the mangrove forest was developing, the salt marsh would yield many of the same ecosystem benefits. The marsh-to-mangrove community would also contribute indirectly to nearby seagrass recovery through improved water quality (reducing nutrient and sediment loading), thus benefiting a wider estuarine ecosystem restoration effort. Because the trustees recognized that short-term ecological injuries and service losses could occur during construction activities they specified the use of booms and other various control mechanisms to minimize disturbance. Specific details on scaling the services provided by the newly created habitat to match the mangrove injuries are not available in the case document (FDEP *et al*., 1997). The delivery of all ecosystem services of a fully functional mangrove habitat that began as a rudimentary salt marsh would not occur as rapidly as the full restoration of nekton production that was projected after only 5 years in the Fort Lauderdale mystery spill (NOAA and FDEP, 2002) mangrove restoration.

2.4.3.2 Service-to-service scaling: mangrove restoration to replace lost fish/invertebrate production

In the Fort Lauderdale mystery spill (NOAA and FDEP, 2002) case, the trustees chose the creation of mangrove forest habitat to compensate for lost production of fish and invertebrate biomass that comprised the water column injury. Literature data on the secondary production of mangrove forests were used to scale the number of acres of habitat needed to match the lost faunal production. Field studies by Yanez-Arancibia *et al*. (1980) showed that southern Gulf of Mexico mangrove habitats produce 12 g of fish and invertebrates m^{-2} y^{-1}. Consequently, each acre of fully functional mangrove forest would produce 1,091 kg annually. To account for the trajectory of increasing production over time for a newly created mangrove forest, trustees employed literature-based estimates of recovery rates of fish and shrimp in restored mangrove forests. The abundance of these nektonic animals in restorations reached equivalence with natural mangroves within 5 years. Scaling assumed that this service of newly restored mangal habitat would increase linearly and reach full functionality in 5 years, with persistence for 50 years. Thus the nekton production service benefits of each acre of mangrove restoration were scaled accordingly to compute area of restoration needed for full compensation of the injury, accounting for time elapsed between injury and replacement with the usual 3% discounting. No subtraction was included for loss of ecosystem services of the upland habitat that must be cleared and graded to prepare for mangrove introduction because the sites considered for this restoration project benefit by the removal of nuisance exotic plants at densities of 30-65% cover in one site and 90% cover in another. Creation of a mangrove forest also serves other resources, like wading birds that use the structure for roosting and nesting, so this set of contingent services also helps balance the failure to subtract for lost services in the uplands.

2.4.4 Seagrass restoration

Seagrass habitat resembles salt marsh, mangrove forest, and oyster reef in representing a biogenically structured habitat in which the organism protrudes from the sediment surface with consequent influence on bottom flow regime, the geochemistry of the surface sediments, and the animals that occupy the habitat. Like these other biogenic habitats, seagrass habitat has suffered dramatic declines in most estuaries of the U.S. and the world (Lotze *et al*., 2006). Seagrass is highly productive, induces production of benthic microalgae, enhances production and abundance of benthic invertebrates, and supports abundant nektonic fishes and mobile crustaceans. Consequently, seagrass beds represent a logical target for compensatory restoration. Numerous small seagrass restorations occur as primary restoration and even compensatory restoration for small injuries from boat groundings. There has been some reluctance to conduct large-scale seagrass restorations in part because of concern over the longevity of the created seagrass bed, even under circumstances where the restoration is located on a site known to possess seagrass historically. In addition, unlike *Spartina alterniflora*, no seagrass plants are available for transplanting from nurseries, so seagrass restoration requires excavation of plants or plugs of sediment holding plants from a healthy seagrass bed. This procedure raises concerns over the injury caused by extraction of seagrass.

In areas where the life histories of the injured local seagrasses include good colonizing abilities, seagrasses may recover naturally after perturbations (Preen *et al*., 1995). Recolonization of seagrasses into denuded patches occurs more readily by vegetative spread than by natural reseeding, so smaller areas of injury that possess neighboring undamaged seagrass are more likely to be candidates for natural restoration. Absent active restoration, however, the severity of the environmental impact or the involvement of a species with an extremely slow recovery rate may lead to long-term loss of seagrass habitat (Fonseca *et al*., 2002). Once seagrass cover is removed, a negative feedback is induced that inhibits natural recovery. As seagrass canopy disappears, the canopy's effect of baffling the current flow also ends, leading to increased flow speeds and higher bottom shear stresses. This erodes the finer sediments typical of seagrass beds, stops the process of fine sediment deposition, and modifies the bottom habitat, creating an environment that is erosive not depositional, sandy rather than muddy, and in which turbidity becomes elevated. High bottom shear stress even inhibits settlement of seagrass seeds, making natural recolonization difficult. Seagrass beds and sand flats may on some long time scales be considered alternative persistent states of the same community (Peterson, 1984). Consequently, for injuries to slowly colonizing seagrasses or where the injury involves denuding of large patches, active restoration is typically required to bring back the seagrass habitat and its associated ecosystem services. Even when restoration is selected, physicochemical properties of the seagrass environment must be established before successful restoration of seagrass habitat can occur (Fonseca *et al*., 2002). Seagrass bed restoration can be used not only to replace lost seagrass or some other habitat loss but also to replace animal losses by providing trophic support. The scaling of such restorations of habitat in kind, as a conversion from another injured habitat, or to compensate for lost production services of a guild of seagrass-occupying animals follows the same conceptual model as the salt marsh scaling, with parameter details changed to account for biological differences between the salt marsh and seagrasses.

2.4.4.1 In-kind compensation

The Mini-312 Program (as described in Section 2.2.8 on seagrass injury assessment) describes the process of scaling a seagrass habitat restoration project (HEA). Scaling in-kind HEAs is very straightforward in that no conversions are necessary and the only units are acres of seagrass habitat. This procedure assumes that seagrass restoration projects will reach maturity at 100% of natural seagrass bed services in five years, have a linear recovery curve, and provide services in perpetuity (Kirsh *et al*., 2005). Successful seagrass restoration projects have shown that the relative productivity of restored seagrass habitat is equal to that of natural habitat (Fonseca and Bell, 1998). The spatial scope of the restoration project is scaled in the standard fashion with discounting to offset the interim losses associated with the injury for which it is compensating (Fonseca *et al*., 2000). Once the seagrass injury is established in units of m^2-years of seagrass ecosystem services (following the Mini-312 Program protocol), the $m^2$of seagrass to be restored is easy to calculate by dividing total injury in m^2-years by the number of m^2-years of services provided by one m^2 of restored seagrass (e.g., Kirsh *et al*., 2005). Most seagrass habitat restorations are located in "orphan" injury sites, defined as injured seagrass beds lacking identification of the responsible party and thus lacking financial support for primary restoration. In examples where an orphan injury site is used for compensatory restoration, the benefits of the restoration in acre-years of seagrass habitat services must be computed by subtracting away the growth in service acre-years from unaided recovery expected in the absence of any intervention. In cases in which the recovery trajectory of a target species is known to be relatively slow like *Thalassia testudinum*, a native pioneering species (*e.g*., *Halodule wrightii*) can be used to facilitate and speed up the eventual recovery of the slow-growing species that must be restored for full compensation. This facilitation has been termed "compressed succession" (Moffler in Fonseca *et al*., 2000). Fonseca *et al.* (2000) provide practical and informed guidance to successful seagrass restorations.

2.4.4.2 Scaling conversion of injuries in one habitat to seagrass habitat restoration

We were unable to find any cases in which seagrass habitat was to be restored to compensate for injury of another type of habitat. Nevertheless, such habitat conversions would be based on the same rationale used to convert various habitats into restored salt marsh. Such conversion scalings are illustrated in French McCay and Rowe (2003), in computations of how injuries to various guilds of aquatic animals following the North Cape oil spill could be compensated quantitatively by the trophic support function of restored seagrass (eelgrass) habitat. French McCay and Rowe (2003) develop scaling approaches based upon productivity metrics for habitat services. Expected gains in production from restoring seagrass on a shallow unvegetated sedimentary habitat were converted to production benefits at multiple trophic levels, dictated by the trophic level on which injury occurred. This trophic enhancement scaling was computed using each of two different empirical data bases: (1) primary production assessed by local empirical data and (2) secondary production of benthic invertebrates also based on local field data. Empirical data on eelgrass (*Zostera marina*) productivity from a Rhode Island salt pond yielded annual production of 1429 g dry weight m^{-2}. Benthic microalgal production data within a seagrass bed habitat come from a Chesapeake Bay study (Buzzelli *et al*., 1998): the production of microalgae on plant surfaces and on sediment between plants totaled 400 g dry weight m^{-2} y^{-1}. Because the seagrass restoration would be conducted in and replace unvegetated shallow subtidal flat habitat, the net gain in primary production of microalgae is computed by subtracting the primary production of a shallow unvegetated flat (assumed to be 124 g dry weight m^{-2} y^{-1} from a South Carolina field assessment: Sullivan and Currin, 2000) from the 400 g total, yielding 276 g dry weight m^{-2} y^{-1}.

To convert these estimates of primary production gains to enhanced production of benthic invertebrates, which are comprised of herbivores feeding on microalgae and detritivores feeding on microbes on seagrass detritus, French McCay and Rowe (2003) applied an ecological efficiency of 6.6% for the conversion of seagrass to detritivores through microbial mediation identical to what Kneib (2003) describes for salt marsh detritus. The efficiency of herbivory on benthic microalgae is assumed to be 10%. Summing these pathways yields a net gain in secondary production of benthic invertebrates of 122 g dry weight m^{-2} y^{-1}. This figure compares to a net difference in sampled benthic invertebrate production between eelgrass and shallow unvegetated bottom of 175 g dry weight m^{-2} y^{-1}, a figure not too different from the enhancement of secondary production projected from knowledge of primary production differences. Scaling restoration to replace lost production of secondary producers in a Rhode Island salt pond can thus be computed with some confidence using an enhancement of restored seagrass habitat over the shallow subtidal habitat it replaces of between 122-175 g dry weight m^{-2} y^{-1}. To conduct scaling for replacement of nektonic fishes and crustaceans at the third trophic level can be based on assuming a 20% ecological energetic efficiency in converting secondary production to tertiary (French McCay and Rowe, 2003).

As in all other scaling applications, the complete scaling exercise requires time-varying factors to be included in integrating the benefits of restoration over the lifetime of the project. The project computations for the North Cape seagrass scaling was assumed to last 100 years under the presumption of protection of the site and management to maintain the bed if necessary. Based on the review of seagrass restoration projects by Fonseca *et al*. (1998), the restored seagrass bed was also assumed to provide 99% of the services generated by a natural seagrass habitat and to reach this level in 3 years, increasing linearly. Finally, discounting is applied in the time integration to account for public preference for immediate service over delayed provision.

Because French McCay and Rowe (2003) scaled the lost production of benthic invertebrates, nekton, pond birds, and other guilds of consumers to both salt marsh and seagrass bed restoration, comparisons between the acres required for these two types of restoration represent the habitat conversion factors between salt marsh and seagrass habitat. The habitat conversion factors vary depending on whether the scaling was done on the basis of primary productivity or secondary production data, but in both cases the seagrass habitat is far more valuable per unit area. Using the primary production metric, the conversion factor specifies that an acre of restored seagrass is worth 7 acres of restored salt marsh, whereas the ratio is 11 when computed using comparable data on secondary production of benthic invertebrates. These ratios seem unreasonably high. Some of the cause is that salt marsh is estimated to require 15 years to reach the maximum delivery of services as opposed to the 3 years of development for restored seagrass beds. The assumption of a 100-year life span of the project would be very unrealistic for seagrass without active management, and may be a serious overestimate even with management because seagrass is sensitive to water quality deterioration and stormwater controls seem broadly inadequate to protect against estuarine turbidity, sedimentation, and eutrophication (Lotze *et al*., 2006). A more recent synthesis of empirical data on habitat-specific productivity at each of the three lowest trophic levels (Peterson *et al*., 2009) suggests that seagrass and salt marsh possess approximately equivalent levels of trophic production services; however, incorporating the much more rapid development time for restored seagrass than for created salt marsh would lead to greater benefits per acre for seagrass after conducting the necessary time integration. Peterson *et al*. (2009) report that seagrass habitat provides 2-3 times the production services of intertidal flats and 3-4 times shallow subtidal flats, but that oyster reef habitat outproduces seagrass by a factor of 3-8.

2.4.5 Coral reef restoration

Coral reefs are universally recognized as a significant habitat providing structural heterogeneity and serving as refuge for an astoundingly diverse suite of sessile and mobile organisms. The foundation of a coral reef ecosystem is a web of symbioses between the coral animal and zooxanthellae, microalgal symbionts held within the tissues of hermatypic corals, and between the calcium carbonate of coral skeletons and many other invertebrate and algal producers of calcium carbonate. The coral reef ecosystem is known for its complex web of interdependencies and for its dire status under threats of a warming, more acidic, and higher sea (Hoegh-Guldberg *et al.,* 2007). The many ecosystem services of coral reefs generate tremendous social, cultural, and economic value, ranging from support of subsistence of many indigenous populations to support of massive flows of international tourists on developed coasts. Consequently, any injury to a coral reef should induce an effective in-kind restoration to compensate for losses to such a valuable habitat. Unfortunately, coral growth rates tend to be very slow, complicating restoration and potentially requiring large areas of restoration to match injuries. The natural rate of coral reef expansion in the Florida Keys is only 0.65–4.85 m per 1000 years (Shinn *et al.,* 1977). Individual coral species grow at rates that range from about 1 cm y^{-1} for massive head corals to 10 cm y^{-1} for "weedier" branching corals (Jaap, 2000). Corals require an open hard substrate on which to recruit and their recruitment is facilitated by coralline algae. Because reef expansion and coral growth rates are relatively slow, and natural recruitment rates are highly variable over time, in space, and among species, even under optimal conditions, many clever interventions are useful to accelerate compensatory coral reef restoration.

2.4.5.1 In-kind restoration - installation of artificial reef modules

Coral reef service losses resulting from physical damages to corals from the *Berman* grounding case (Tetra Tech, 2006) were scaled to coral reef creation. Pre-fabricated, cement modules were deployed in different orientations to serve as the structural matrix for a new coral reef. This artificial reef matrix provided necessary hard substratum for the natural recruitment of corals and other sessile reef biota and subsequently suitable habitat for other reef-dwelling organisms. Restoration scaling assumed that the reef developing on this artificial core would require 50 years to reach maximum productivity on a linear recovery trajectory. Maximum production was projected to reach only 75 % of the ecosystem services of a similar two-dimensional surface area of natural reef at the grounding site before injury because the restoration was sited in deeper waters where lower productivity was expected based on light limitation of the zooxanthellae in the largely hermatypic corals. Because this provides in-kind compensation, scaling could be done to effective reef surface area, defined as two-dimensional surface area open to flow, exposed to light, at proper depths, and thus suitable for occupation by corals and other reef epibiota. Ecosystem services from the artificial coral reef were expected to last indefinitely. Site suitability analyses allowed choice of a location that allowed confidence in the scaling projections.

This *Berman* scaling failed to account for the sand bottom habitat that was lost when the reef module covered it. Such losses of the habitat that is replaced by a restoration need to be included in future coral reef restoration scaling computations. In addition, the coral reef injuries for which compensation was computed in the *Berman* case represented only that reef area that was physically damaged by the hull during grounding. There was also release of 800,000 gallons of No. 6 fuel oil from the grounded barge. No sublethal injury to coral reef habitat from this contamination was computed or included in the scaling of compensatory restoration. In a more recent case of an estuarine oil spill, the trustees computed oyster reef injury based upon percentage reduction in growth rate (and thus production) of oysters (see oyster reef injury section). Corals are likely to be similarly affected by exposure to oil and such reductions in production should be considered injuries that deserve inclusion in the restoration scaling.

2.4.5.2 Monitoring and modeling coral reef recovery time on artificial reef modules

Lirman and Miller (2003) monitored restoration progress (coral species richness, and density and colony diameter of the dominant coral) on the restoration sites created after the grounding of the *R/V Alec Owen Maitland* and *R/V Elpis* in the Florida Keys National Marine Sanctuary (FKNMS) and compared these data with outputs of a simulation model. Both groundings had occurred in fall 1989, with restoration projects initiated in summer 1995 near their respective grounding sites and monitoring surveys conducted at 3 and 6 years post restoration. The *Maitland* restoration project employed interconnected concrete slabs with low topographical relief, whereas the *Elpis* restoration employed limestone boulders that produced high relief. Coral recovery depended on natural recruitment at both sites. The relatively opportunistic coral

Porites astreoides was chosen as the chief metric to gage recovery because of its dominance on the restorations and natural reference areas. Haphazardly placed belt transects were used to sample abundances of each of five size classes. These size classes were used in a simulation model (a size-structured transition matrix model) to evaluate convergence rates of coral population structure and abundance between restoration sites and adjacent reference habitats.

The stage-based population model for *P. astreoides* inputs coral life-history attributes (colony growth rate, recruitment, and clonal mortality and fragmentation) and provides trajectories of convergence in coral colony abundance and size structure. The model was run at multiple recruitment rates (0.15, 0.30, and 0.45 recruits m^{-2}). Within 6 years, 70-80% of coral species found on reference sites had colonized the restorations. Abundance of *P. astreoides* was indistinguishable from nearby reference areas at both sites within 6 years; however, coral colony size distributions remained different with smaller colonies at the recovery sites. From the third until the sixth year, coral abundance at the low-relief *Maitland* restoration site doubled while remaining unchanged at the *Elpis* restoration site. Simulation modeling revealed that *P. astreoides* abundance and size structure was expected to converge with reference habitats after 10 years but that maximum coral size would require 20 years to recover. Results of the simulation model were sensitive to recruitment rate, highlighting the special importance of recruitment on the recovery rate for coral reef restorations. This modeling involved an opportunistic species with high recruitment rates and thus cannot be extrapolated to less opportunistic species like *Montastraea* spp. and *Acropora* spp., which are likely to recover substantially more slowly (Gittings *et al.,* 1990, as presented in Lirman and Miller, 2003). Thus, for an injured coral reef initially dominated by more opportunistic corals, assumption of a 50-year period to reach maximum coral reef services, as used in the *Berman* case, may be excessive, as judged by the metric of population recovery of the dominant coral; however, recovery of coral communities on a reef dominated by the long-lived massive corals is likely to require far longer than the 20 years for *Porites astreoides*. Progress is being made in artificial propagation of corals in the laboratory: sources of juvenile corals for transplantation, especially for those that recruit so slowly in nature, could greatly reduce times required for restoration and recovery of coral reef services.

2.4.5.3 Restoration by coral transplantation

Graham and Schroeder (1996) measured the success of reattaching hard corals on a nearshore reef as a possible restoration technique after the grounding of the *M/V FIRAT* near Ft. Lauderdale, Florida. The *M/V FIRAT* lost its anchorage during a storm. Thereafter, its hull scraped across the limestone-ridge hard bottom, dislodging hard corals and other epibiota. This incident presented an ideal opportunity to test transplantation success because it generated a ready source of living but detached hard corals so no disturbance of donor areas was required. The hard coral colonies included specimens obviously dislodged during the grounding and perhaps other specimens dislodged by natural processes like bioerosion. The brain corals *(Diploria* spp.), star coral *(Montastrea cavernosa),* and boulder coral *(Dichocoenia stokesi)* comprised the majority of coral colonies available. A total of 588 hard coral colonies was reattached using marine epoxy or bolts (for larger colonies) at 16 coral reattachment areas within the injury site. Of the 588 reattached coral colonies, 127 were permanently tagged and mapped for follow-up monitoring. Specimens of 12 species of hard corals were reattached. After five months, approximately 90% of the reattached coral colonies were judged healthy and firmly attached to the substratum.

Milon and Dodge (2001) provide a broader context to the importance of the ability to reattach and thereby sustain dislodged corals. They point out that investing money in successful primary restoration of the injured reef site may result in substantial cost savings because it reduces the scale of compensatory restoration required later. For physical injuries to coral reef habitat, reattachment of available coral colonies may be a feasible means to limit interim losses of reef services for which compensatory restoration is required. Milon and Dodge (2001) caution those applying HEA to coral reef habitat injuries that the linkages between documented injuries to ecological reef structure and to reef functional services and then to provision of value to humans is not well established and need further study to support confident HEA applications. This issue of human valuation affects especially the time periods over which recovery and restoration occur because reef structural metrics, provision of ecosystem services, and support of human values are likely to follow different time courses in both recovery and restoration. This argues perhaps for merging ecological and human service metrics in coral reef HEAs.

2.4.5.4 HEA scaling software

Kohler and Dodge (2006) introduced "Visual HEA" software using an example in which compensatory restoration of a coral reef was scaled for injuries caused by a planned beach nourishment project. The software enables users to modify a series of variables in order to evaluate hypothetical scenarios for alternative restoration strategies. The input variables include baseline-level of ecosystem services (pre-injury), discount rate, year of claim, parameters that determine service loss from the injury, parameters that determine service gain from the compensatory action, and relative value of pre-injury services and compensatory services at equilibrium. The input parameters for the HEA software are listed in Table 2.8 (Kohler and Dodge, 2006).

Table. 2.8. Parameters for Habitat Equivalency Analysis (Kohler and Dodge, 2006)

Time variables	
$t = 0$	time when injury occurs
$t = B$	time when injured area recovers to baseline levels
$t = C$	time when the claim is presented
$T = I$	time when the habitat replacement project begins to provide services
$t = M$	time when the habitat replacement project reaches full maturity
$t = L$	time when the habitat replacement stops yielding services
Value variables	
Vj	value per area-time of services provided by injured habitat
V^p	value per area-time of services provided by replacement habitat
X^j_t	level of services provided by injured habitat at end of time t
b^j	the pre-injury baseline level of

	services per area of injured habitat
X^p_t	level of services provided by replacement habitat at end of time *t*
b^p	initial level of services per area of replacement habitat
ρ_t	discount factor, where, $\rho_t = 1/(1+r)^{(t-C)}$ r=discount rate per time unit
J	number of injured area units
P	size of compensatory replacement project
Calculated quantities	
$(b^j - x^j_t)$	extent of injury at time t
$(x^p_t - b^p)$	increment in services provided by replacement project
$(b^j - x^j_t) / b^j$	percent reduction in services per area for injured area, relative to the injury site baseline level of services
$(X^p_t - b^p)/ b^j$	percent increase in services per area for replacement site, relative to the injury site baseline level of services

The selected case study in Kohler and Dodge (2006) was simplistic in its conceptualization. It does, however, illustrate with clarity the potential benefits and disadvantages of various restoration options. Assignments of recovery time of the injured habitat and time for development of ecosystem services of the restored habitat are revealed to be critical to HEA calculations. Through a graphical interface, the Visual HEA software allows users to quickly alter these and other parameters as a form of sensitivity analysis.

2.4.5.5 Prevention of future injuries as compensation for coral reef injuries

In February 1997, the *Contship Houston*, a 660-foot container vessel, grounded in the lower Florida Keys, resulting in major damage to 7,663 m^2 and lesser scraping of 662.5 m^2 of coral reef habitat. Injury assessment suggested a 100% loss of services to 938 m^2, a 61% loss of services to 6,725 m^2, and a 25% loss of services to 662.5 m^2. Primary restoration was quickly initiated; however, the natural resources trustees and the responsible party cooperatively agreed that return of full function of coral reef ecosystem services would be realized only after several decades. Hence, compensatory restoration was required. Although details of scaling the resource injury are not readily available (e.g., a recovery trajectory for services is missing), the compensatory restoration project of choice was installation of a RACON navigational system. HEA was applied to match the coral reef injury to the amount of coral reef damage by future vessel groundings that would be avoided by operation of the RACON navigational system (Chapman and Julius, 2005). The decision to scale coral reef injury to an injury prevention project was made only after it passed several criteria: (1) the preventative project used to compensate for habitat loss must provide services that closely match those of the injured natural resource; (2) the preventative project must have a reasonable probability of success; and (3) the preventative project must be consistent with all major NRD mandates - restore, rehabilitate and/or acquire the equivalent of natural resources (Chapman and Julius, 2005). The justification for using compensatory restoration funds for a preventative response is debatable. The immediate and lasting consequence is the loss of services from that habitat because the injury is not replaced.

The assumption in making this choice is that management of the stressors to this habitat is so imperfect that future losses without compensation from a responsible party are inevitable, so prevention ensures more functional coral reef habitat than there would otherwise be at some time in the future. That outlook reflects a rather depressing position on management or our coastal resources.

2.4.5.6 Global climate change impacts

Overwhelming evidence now suggests that increasing atmospheric carbon dioxide and other greenhouse gas levels have resulted in a substantial change in the Earth's climate (IPCC, 2007). The continuing growth of CO_2 poses growing challenges to coral reef habitat via three separate stressors: (1) increasing mean global sea surface temperatures; (2) increasing eustatic sea level; and (3) decreasing oceanic carbonate-ion concentrations, causing ocean acidification (Kleypas *et al.,* 2006; Hoegh-Guldberg *et al.,* 2007). Sea temperatures are warmer, and pH and carbonate-ion concentrations lower than at any other time during the past 420,000 years, all related to increased atmospheric carbon dioxide levels (EPICA, 2004; Hoegh-Guldberg *et al.,* 2007; IPCC, 2007). The rate of change in these parameters over the past 100 years is 2-3 orders of magnitude greater than the mean change in the past 420,000 years (EPICA, 2004), and exceeds limits to which many reef organisms can adapt (Hoegh-Guldberg *et al.,* 2007). Hoegh-Guldberg *et al.* (2007) suggest that changes in global climate are likely to result in the loss of a large proportion of the Earth's coral reefs. Worldwide coral reefs have already declined in habitat area by about 50% over the past 30 years (Bruno and Selig, 2007).

The current impacts of climate-related coral reef stressors are most evident as coral bleaching events. Coral bleaching refers to the loss of color of corals due to the stress-induced expulsion and/or reduced photosynthetic pigment concentrations of the symbiotic zooxanthellae that live within coral tissue (Kleppel *et al*., 1989). Large-scale bleaching episodes are usually associated with unusually high sea surface temperatures (Brown, 1997; Jones *et al.,* 1998; Carpenter and Patterson, 2007). Ocean acidification and decreased carbonate-ion concentrations inhibit calcification of marine organisms, which may limit new skeletal growth in corals and reef-building calacareous algae that the coral reef may be unable to grow vertically at the rate required to keep pace with rising sea level. These catastrophic risks to coral reef habitat suggest that novel means of preventing further climate-related injury to coral reef habitat may soon be the most effective restoration approach.

2.4.6 Kelp forest restoration

Because kelp forests provide a biogenic structural habitat rising up many meters above the shallow rocky seafloor and produce substantial plant biomass as well as providing structural habitat benefits to other plants and associated invertebrates, fishes, and marine mammals (Witman and Dayton, 2001), kelp forest creation represents a reasonable option for use in compensatory restoration or mitigation. Sufficient numbers of quantitative studies of kelp and associated invertebrate and fish resources have been conducted to allow estimation of benefits and thereby to provide a basis for quantitative scaling of benefits of kelp forest creation (see Witman and Dayton, 2001 for a review of concepts and bibliography). An example of scaling methodology can be found in the research reports done to assess the impacts of the San Onofre

Nuclear Generating Station's use of ocean water for cooling (SONGS, 1989) and the mitigation benefits of kelp forest creation (Ambrose, 1990).

The SONGS (1989) study is believed to provide the most comprehensive quantitative study of a kelp forest habitat and its associated invertebrates and fishes conducted anywhere in the world. This study quantified losses in area of the kelp bed, in standing stock biomass of fishes, and in abundance of large benthic invertebrates that resulted from occasionally enhanced turbidity when the discharge plume from the diffusers of a nuclear power station was transported over the San Onofre kelp bed and from the deposition of anomalous fine sediments on the rocks underneath the kelps. By developing and applying the BACI (Before After Control Impact) statistical test method of examining how the relationship between the impact kelp bed and a paired control kelp bed changed from a multi-year period before to a multi-year period after operation of the diffusers, quantitative impacts of plant operations on the San Onofre kelp bed and its major associated animals were determined. Operation of these diffusers for Units 2 and 3 of the nuclear power plant resulted in loss of 80 hectares of kelp forest habitat, as defined by medium-to-high densities of kelp plants (> 4 plants per 10 m^2 (SONGS, 1989). The habitat area represented the variable on which compensatory scaling was then computed (Ambrose, 1990). This approach assumes that abundances of large kelp-associated invertebrates and biomass of bottom and mid-water fishes will follow in proportion to habitat area.

To restore the 80 hectare loss of kelp forest habitat, the restoration scaling specified construction of 120 hectares of artificial kelp reef. The additional area of rocks was intended to account for potential burial of the cobbles and boulders installed to construct the reef substrate and for the likelihood that kelp would not cover all even suitable substrate (Appendix Tech. Rep. H to SONGS, 1989). Assumptions included design criteria: (1) siting the new kelp forest on suitably firm sand and rock rubble bottom to minimize chances of sinking; (2) locating the new reef where sand deposition and scouring are unlikely; (3) utilizing cobbles and boulders in a vertical relief that resembled that of the affected kelp bed and provided some vertical protection against burial; (4) placing the constructed reef outside the influence of the diffuser plume and other potential negative influences; and (5) monitoring the progress of establishment of kelps and associated resources so that adaptive management could be conducted.

This choice of a 1.5 multiplier ratio follows similar practice to account for uncertainty in restoration success in mitigation for losses of other habitats and would have been larger except for the commitment to monitoring and adaptive management of the restoration. If after 3 years, 90% of the 120 hectares did not remain as emergent cobble and boulder rock surface, then additional installation of rock was required. If after 3 years, at least 60% of the reef was not covered by moderate-to-high densities of adult kelp plants, then active kelp planting would be triggered. Similarly, if after 10 years, required monitoring showed a failure to attain a standing stock of fishes of 28 tons (equal to the fish injury) with similar diversity, reproductive rates, and young-of-year densities, then further adaptive management would be mandated. Likewise, if the monitoring of large benthic invertebrates did not demonstrate replacement of the total abundance and native diversity of benthic macroinvertebrates and algae after 10 years to match the injury loss, then adaptive management would also be required. Furthermore, any undesirable invasive species would require attention and control. The inclusion of sufficient financial resources to conduct such detailed monitoring and support any necessary adaptive management represents ideal assurance that restoration will be scaled to match loss of the most important ecosystem services. However, this goal is typically not achievable in most settlements of NRD, CERCLA, and OPA cases in which responsible parties do not accept the uncertainties of unspecified future obligations. Settlements with responsible parties in NRD, CERCLA, and OPA cases may include a contingency to handle monitoring and fine-tuning deemed necessary, such as a fixed 10% augmentation of the dollar costs of restoration. Such an approach is not likely to guarantee full quantitative restoration. This more extensive monitoring and adaptive management approach was viable for SONGS because this case represented mitigation for a perpetual injury under permits that could be re-issued and re-conditioned. In addition, the SONGS study and restoration fell under careful scrutiny from the court, which dictated the terms of impact studies as part of a case settlement. In cases lacking sufficient commitment to monitoring and adaptive management of restoration projects, the 1.5 multiplier would need to be replaced by a larger number. The SONGS restoration scaling also failed to include any discounting for the several years of interim losses between injury from operations and complete quantitative replacement of that injury through restoration. Inclusion of such discounting would be straight-forward and is necessary to satisfy current policy reflecting the higher public valuation of present than future resources.

2.4.7 Habitat preservation and conservation easements

Several projects that have been considered compensatory restoration for injuries arising from environmental incidents function by preventing future injury rather than by intervening to enhance resource or habitat levels. These protection projects are justified as compensatory restoration on the grounds that the recent past history of how human activities have increased and impacted specific resources or habitats implies that future resource or habitat loss and degradation is inevitable. Such logic is an indictment of the efficacy of existing environmental protections, yet probably serves as a realistic projection of future losses and thus renders protection a legitimate compensatory restoration action. However, such applications of restoration should be restricted to cases where no compensatory restoration is anticipated for future injuries, where it is clear that no improvement in environmental management will emerge to provide the protections anyway, and where quantitative scaling information exists on which to relate the amount of protection achieved by the project to the quantity of resource or habitat protected. Conservation easements, purchases of development rights, and other actions represent examples of preservation that could be used to provide compensation or mitigation for some environmental injuries.

A model example of using land purchase as compensatory restoration for loss of a wide suite of injured species comes from the North Cape oil spill case in coastal Rhode Island. The land purchase of building lots in close proximity to a salt pond was intended to protect, from storm-water run-off of nutrients and sediment, eelgrass habitat that falls under current environmental protection rules but nonetheless is being degraded as an inevitable consequence of the development of housing (French McCay and Rowe, 2003). The ability to use this project for quantitative restoration depended upon the availability of a relevant credible study (Short and Burdick, 1996) relating the numbers of houses in the watershed of Ninigret Pond to loss of eelgrass acreage: specifically, each added house resulted in loss of 1300 m^2 of eelgrass habitat. Knowing this relationship developed for one of the ponds that were choices for implementing the protection action then allowed scaling to match the injury. Injury to production foregone of benthic macro-invertebrates, pond birds, and small and large fishes (French McCay and Rowe, 2003) could then be quantitatively related to production benefits provided per unit area of eelgrass habitat using HEA knowledge of production in eelgrass. French McCay and Rowe (2003) computed these production benefits in units of both primary production and secondary production of benthic macro-invertebrates, which yielded somewhat different answers (21 vs. 30 acres required). This approach can be used for other projects that might mitigate or future injuries from storm-water run-off. Other BMPs like establishing vegetated buffers, limiting impervious surfaces, or restoring or maintaining natural hydrography may also be scaled to provide a viable option to protect against future injury from development.

2.4.8 Preservation of nesting habitat for birds

Assuming that injury assessments following environmental incidents have measured bird losses in units of bird-years lost, then quantitative compensatory restoration actions can be scaled to match the injuries. For many species of birds, especially endangered and threatened species, protection, preservation, or restoration of nesting habitat represents a likely method of enhancing bird population abundances. Protection commonly involves isolating the breeding birds from human activities that are currently disrupting breeding and chick rearing. Preservation of breeding habitat will not, in contrast, augment the targeted bird population, but it will inhibit future decline. If that future decline can be shown to be inevitable under assumptions of continuing present management regimes, then prevention of this disruption can be arguably considered a compensatory restoration. More compelling is completion and maintenance of a compensatory restoration or mitigation project that restores degraded nesting habitat for birds that are now limited in part by suitable nesting habitat. Here we present a methodological overview of examples of each approach: nesting habitat protection (Donlan *et al*., 2003), preservation (Sperduto *et al*., 2003), and restoration (NOAA *et al*., 2002).

2.4.8.1 Nesting habitat protection as compensatory restoration

Quantitative restoration or mitigation for threatened or endangered species is facilitated by the existence of agency-prepared species recovery plans based upon comprehensive study of the threats limiting the populations of these species. Such a plan applies to piping plovers, one of the species injured by the North Cape oil spill. That plan had led trustees for a previous oil spill, the World Prodigy in 1989, to choose a restoration project that extended annual seasonal protections against human disturbance to several potential plover nesting areas: coastal barrier sites suitable for plover nesting (low vegetation density, access to marine benthic invertebrate foods, etc.) but not currently used by plovers because of human disturbance. New nesting sites like these are more likely to be adopted by young adult (new) breeders, which can be induced by excluding humans with warning tape and signs. Consequently, the trustees for the North Cape oil spill had benefit of being able to analyze the effectiveness of this previous plover restoration project in scaling their own analogous project to be able to quantitatively match observed injury (Donlan *et al*., 2003). After establishment of these same protections at EB Watch Hill, breeding plovers produced 11 chicks, when no more than 2 had been fledged in the three previous years. Unfortunately, despite this evidence of success, the general effectiveness of the habitat protection measures applied after the World Prodigy varied in space and over time, reducing confidence in quantitative scaling to replace the 5 chicks lost to the North Cape spill. This uncertainty is motivation for monitoring the effectiveness of the restoration project, the results of which can be applied to adjust the scale upwards if necessary restoration is not fully achieved. This flexibility for adaptive management of restoration projects represents an important component of the process, requiring resources for monitoring and modifications.

2.4.8.2 Nesting habitat preservation as compensatory restoration

To compensate for the loss of 3286 loon-years, trustees for the North Cape oil spill chose to preserve and protect existing loon nesting habitat. This option was selected over alternatives of nest site enhancement and public education because the high uncertainties in these latter two alternatives prevent confident quantitative scaling. Preservation of loon nesting habitat was contemplated in the form of purchase of lands or development rights on the shores of freshwater lakes where loons are known to nest. The quantitative scaling involved first determining the average productivity benefit of preventing development, measured as per nest enhancement in fledgling production. In addition, the benefits of preventing development included enhancing survivorship of adult birds, which was also quantified by using data comparing developed and undeveloped lakes in a New Hampshire nesting region. This information was combined with knowledge of demographic parameters of age-specific survival and reproduction for loons to quantify benefits from augmented productivity of both first- and second-generation beneficiaries repeated over an expected 100-yr project life time. In principle, the preservation of nesting habitat may be indefinite, but with inclusion of discounting at 3% per annum the contributions after 100 years are negligible.

So reproduction preserved by preventing nesting area development was computed for first-generation breeders and second-generation breeders separately. Then the two were added to provide the full per nest productivity benefits. The first-generation gain per nest in loon-years was computed by adding (with discounting) over 100 years the product of the per nest productivity gain (PG: 0.5 fledglings per nest per year) times the average life expectancy of a newly hatched bird (L). The average life expectancy reflected benefits of enhanced breeding success and improved adult survival (by reducing deaths from ingestion of lead sinkers, boat accidents, fishing gear encounters, and plastics) on lakes without development. Similarly, the second-generation gain per nest was computed by multiplying this average productivity gain (PG) by survival of a fledgling to adulthood (FS) by the probability of an adult breeding (pA) by the numbers of adult breeders divided by two (parents of each sex) (PLO/2) times the average life expectancy of a newly hatched bird (L). This second generation term was computed for each year of the 100-year project lifetime, using a 3% discount rate and depreciating the numbers of birds by another term for annual adult survival each successive year. Inclusion in both the first- and second-generation terms of L is required to provide the result in units of loon-years, matching the injury units. Credit was not provided beyond the second generation because of uncertainty over possible action of density-dependent limitations in later years. Summing the discounted values of per-nest productivity enhancement through preventing development yielded the result that about 25 nests required indefinite protection to match the injury of 3286 bird-years (Sperduto *et al*., 2003).

2.4.8.3 Bird nesting habitat restoration as quantitative compensation for injury

To replace the estimated 553 ruddy ducks (adult equivalents) lost from the population in the Swanson Creek vicinity after the Chalk Point oil spill (NOAA *et al*., 2002), trustees chose to scale restoration of a project to restore and then protect nesting habitat for this species. Nesting occurs in a distant location, namely in the Prairie Pothole Region (PPR) of the upper Midwest. The project chosen for quantitative scaling involved restoration of wetland grasses on previously converted farm lands and perpetual protection of that restoration via purchase of easements. Scaling was achieved by applying information on density of ruddy duck nests established in this area (0.038 nests per hectare) and on expected productivity (1.5 adult birds produced per nest per breeding season). The protections of the restoration site supported an assumption of a 100-year lifetime for the restoration. Furthermore, only 70% of the ruddy ducks nesting in the PPR migrates to the Chesapeake Bay, so the credit for compensation was reduced by 30% to achieve goals of evaluating restoration benefits at the injury location. The computations yielded need for 750 acres of restoration of nesting wetlands to provide the required 553 adult ducks. Monitoring was proposed to assess the effectiveness of this restoration and ensure compensation as well as to provide more empirical scientific data for consideration in designing future similar projects.

2.4.9 In-stream improvements for fish

Although our examples to this point have been marine and estuarine, freshwater aquatic systems also suffer impacts from environmental incidents or require mitigation for planned development projects, requiring compensatory restoration or mitigation. Furthermore, rivers and streams have suffered extensive environmental modification and thus deliver depressed levels of ecosystem services. This provides multiple opportunities for restoration of degraded systems and for addressing the sources of ongoing degradation as an indirect means of establishing quantitative compensations. Scaling of such projects is analogous to scaling of brackish and salt-water habitats and resources. Here, we discuss several types of freshwater compensatory restoration scaling, including improvements to spawning habitat for salmon, enhancement of in-stream habitat for trout, and dam removal and fish passage projects whose benefits are measured in terms of improved habitat services for fish.

2.4.9.1 Enhancement of in-stream spawning habitat to compensate for salmon mortality

In many river systems, the quality and extent of spawning habitat is a limiting factor for fish production. In cases involving fish injury, especially anadromous fishes like salmoids, the preservation or enhancement of spawning habitat is likely to increase fish abundance, thus compensating for fish mortality. A common restoration metric is the number of fish gained, measured at a selected life stage (number of juvenile or adult fish, for example). Key inputs include spawning density per unit of restored habitat and life-stage survival rates that convert data on spawning density to the selected life-stage metric. The scaling of fish mortality to spawning habitat improvements has been applied in several NRD cases, including especially the Iron Mountain Mine Superfund site in California (Stratus Consulting, 1999).

In the Iron Mountain Mine case, scaling was performed for two types of spawning enhancement projects: direct in-stream improvements and replenishment of gravel for deposit by river currents. The metric for both projects was increased production of salmon fry (recently hatched fish). Injuries to fish had been accordingly measured as the quantity of salmon fry killed due to contamination from the mine site.

The first project involved modification of the stream channel and the placement of woody debris in the stream bed of a tributary to the Sacramento River south of Redding, California. This type of restoration has been shown to improve habitat for salmon spawning and feeding (Crispin *et al.,* 1993; Giannico, 2000). Restoration scaling relied on the assumption that fish production would double in areas where streambed improvements were undertaken, based on the professional opinion of state biologists who studied the site. The initial level of production was estimated based on state records showing the number of spawning adults that return to the stream each year. It was assumed that half the adults are females, that each female produces a nest of 5,000 eggs, and that 25 percent of the eggs produce fry. Total fry production in the tributary was pro-rated by stream mile to determine baseline production in the 1.9-mile stretch where restoration would take place. Given 203 spawning fish per mile, the estimated increase in annual fry production for the proposed project was 240,000 (203 fish x 0.5 female x 5,000 eggs x 0.25 fry x 1.9 miles).

The second project involved the placement of gravel in selected locations of the Sacramento River below the Keswick Dam in Redding, California. Dams can reduce the extent of spawning habitat by inhibiting the downstream flow of sediment, including gravel. When the supply of gravel is replenished, river currents can deposit the gravel along protected areas of a river channel to help create suitable spawning habitat (Kondolf and Matthews, 1993; Merz and Setka, 2004). Gravel placement programs had been implemented in the same area of the Sacramento River in the past, but were previously discontinued due to a lack of funds. Restoration scaling for the Iron Mountain Mine assessment used the results of those prior efforts along with additional inputs.

An overview of the scaling calculations is presented in Table 2.9. Following the placement of 41,350 cubic yards of gravel during prior restoration efforts, the increase in gravel area potentially suitable for salmon spawning was measured at 537,000 square feet. This effectively doubled the amount of potentially suitable spawning habitat in a stretch of the river less than 10 miles long. State biologists estimated that approximately 10 percent of the potential spawning area would ultimately be used by salmon for spawning. In the areas used for spawning, the occurrence of salmon nests (or redds) was estimated at one per 200 square feet, with about 5,000 eggs per nest. The production of salmon fry was estimated to be 25 percent of the number of eggs. As illustrated in Table 2.9, these assumptions result in a production estimate of 8 fry per cubic yard of gravel deposited in the stream bed.

Table 2.9. Estimated increase in fry production from gravel placement project

Restoration scaling calculations	Units	Result
1. Size of gravel placement	yd^3	41,350
2. Increase in potential spawning gravel area	ft^2	537,000
3. Increase in used spawning gravel area [(2) x 0.1]	ft^2	53,700
4. Increase in number of nests [(3) / 200]		269
5. Increase in number of eggs [(4) x 5,000]		1,342,500
6. Increase in number of emergent fry [(5) x 0.25]		335,625
7. Number of fry per cubic yard of gravel [(6) / (1)]	fry/yd^3	8.1

The gravel must be replenished on an ongoing basis to achieve a consistent level of habitat services in perpetuity. It was determined that the placement of 10,000 to 20,000 cubic yards of gravel annually was appropriate for the target level of restoration at the Sacramento River site. Under steady-state conditions, the gravel placement would just replenish the quantity washed away each year. Ignoring longer-term dynamic effects, the annual gain from restoration was therefore assumed to be 8 fry per cubic yard multiplied by 15,000 cubic yards, for a total increase of 120,000 in annual salmon fry production.

2.4.9.2 Enhancement of in-stream habitat to compensate for trout mortality

The release of metals from mining and mineral processing operations in the upper Coeur d'Alene basin in Idaho have caused a reduction in fish populations in several streams. A decline in the density of trout in contaminated rivers was used as a metric of services losses in a REA model. An increase in trout populations from projects such as channel reconfiguration and the addition of woody debris, implemented on nearby uncontaminated streams, was used to offset losses (Lipton et al. 2004).

Baseline trout densities were estimated using data from upstream locations above the release of contamination from mining operations. Electrofishing data were compiled from upstream reaches on each of the three streams included in the assessment so that stream-specific baseline estimates could be developed. Trout densities per unit area from each reference site were compared to trout densities per unit area from each respective contaminated reach. The percent decline in trout density in contaminated areas relative to reference areas was multiplied by the area of the contaminated stream reaches to estimate the decline in service-acre-years attributable to contamination. Trout densities declined from 0.055 fish/m^2 to 0 fish/m^2, 0.122 fish/m^2 to 0 fish/m^2, and 0.118 fish/m^2 to 0.02 fish/m^2, respectively, on the three impacted streams.

The above estimates related to conditions at the time of the assessment, but planned EPA cleanup actions were expected to reduce future contamination and increase future trout populations in the contaminated streams. To assess losses over future years, the likely decline in annual service losses due to the cleanup actions was estimated based on the future decline in zinc concentrations predicted by EPA analysis. Specifically, a trend relationship was estimated between zinc concentrations and trout densities for numerous sample locations on the impacted streams. The resulting equation was $D_{rb} = -33.5\ln(Z_n/ALC_{Zn}) + 99.7$, where D_{rb} is predicted trout density as a percent of baseline trout density, and Z_n/ALC_{Zn} is the concentration of zinc at the sample location relative to the threshold concentration established in the relevant aquatic life criteria published by EPA for the Coeur d'Alene streams. Using this equation, future trout densities in the impacted streams were predicted based on the future zinc concentrations projected by EPA. Annual service losses for future years were estimated using the difference between baseline trout densities, as described above, and predicted trout densities.

Numerous streams in the Coeur d'Alene basin and nearby river basins are subject to habitat degradation other than metals contamination. Habitat stressors affecting trout populations include channel alterations that reduce habitat complexity, the loss of canopy and woody debris, and sediment input and habitat fragmentation from roads and railways. Restoration projects to address these impacts include restoring channelized streams to a natural meander pattern, relocating roads away from streams and removing abandoned railroad beds, the addition of bank structures to create in-stream pools and provide cover for fish, the addition of woody debris to streams to create cover and reduce erosion, the improvement of culverts that block passage of fish or periodically overflow and introduce sediment to the stream, and other improvements. To estimate the benefits of such habitat improvements, data on trout densities in numerous streams of varying habitat quality were compiled. From the range of trout densities represented in the data, the 20th percentile was used as an estimate of trout density typical of degraded habitat, and the 80th percentile was used as an estimate of trout density typical of restored habitat. The resulting estimate of the increase in trout density from the implementation of the selected habitat improvements was 0.135 fish/m^2. Based on studies in the restoration ecology literature, it was estimated that it would take 10 years after implementation for the habitat projects to produce full benefits. Services were assumed to increase linearly during the 10-year transition. Benefits were assumed to occur over the length of stream reach in which improvements were implemented.

2.4.9.3 Dam removal and fish passage

The mitigation of adverse impacts caused by dams can restore ecological services in rivers and streams. Dam removal and creation of fish passage are the most common types of dam-related restoration. Because large hydropower and flood-control dams can interfere with biochemical and physical properties of a river, restoration actions may also target sediment transport, water temperature and flow conditions. Over 450 dams have been removed in the U.S. during the last century, most of them relatively small structures below five meters in height (Hart *et al*., 2002). Larger dams may also be candidates for removal, but the total number of dams removed in future years is likely to remain a small fraction of the approximately 75,000 dams that exist in the U.S. today (Pejchar and Warner, 2001). Virtually all these dams were built prior to the passage of national environmental laws, but they remain in place in many cases due to important economic and recreational benefits. In some cases dams provide few benefits but funds to remove them are not available.

Numerous published studies have examined ecological changes following dam removal. Hart *et al.* (2002) and Doyle *et al.* (2005) provide useful reviews. The most salient changes following removal of a dam occur in the former impoundment area. In addition to removal of an impassable barrier to fish, observed changes in the impoundment area may include an increase in water flow, an increase in the occurrence of riffles and stream contours, an increase in sediment size, a decrease in water temperature, and deepening of the river channel. Physical changes downstream of the dam may include an increase in sediment transport and a decrease in water temperature. Many of these changes have been associated with ecological benefits, including greater fish diversity, a shift toward more desirable fish species, greater plant colonization and a decrease in fish parasites.

It is possible for the removal of dams to cause adverse ecological impacts. These include the downstream transport of contaminants trapped in the impoundment area, a reduction in wetland habitat or groundwater recharge supported by the impoundment area, and the removal of a barrier to invasive species such as sea lamprey. Dam removal can also cause a change in habitat conditions that adversely impacts certain ecological communities, especially in the short term (e.g., Sethi *et al.,* 2004). Numerous fish ladders and related modifications have been undertaken in the past several decades and they are generally not associated with significant adverse ecological impacts.

Our review of the literature identified several quantitative metrics that could potentially be used in scaling applications. These include an increase in abundance of resident fish above and below the former dam site (Burroughs and Hayes, 2007), an increase in abundance of zoobenthos in the former impoundment area (Casper *et al.*, 2006), and monetary valuation of benefits to recreational anglers from fish passage (Boyle *et al*., 1991). Several other quantitative metrics are available in Bushaw-Newton *et al.* (2002). The first example described below involves an approach to scaling dam removal based on a study in Wisconsin (Kanehl *et al*., 1997). The second example illustrates the use of relative habitat values (Iadanza, 2001; Steger and O'Connor, 2001) to scale a fish ladder project. An example of the enhancement of sediment transport below dams was described in connection with the Iron Mountain Mine case in the previous section on in-stream improvements for fish.

2.4.9.4 Dam removal to compensate for aquatic habitat losses

Using measurements from a Wisconsin dam removal project reported in Kanehl *et al.* (1997), trustees for the Athos oil spill estimated the benefits of dam removal on a tributary to the Delaware River. An increase in habitat quality within the tributary was used to offset spill-related impacts to aquatic habitat in several nearby tributaries.

The Kanehl *et al.* (1997) study reported habitat changes following the removal in 1998 of the Woolen Mills dam on the Milwaukee River in Wisconsin. The Woolen Mills dam was 4.3 meters high and had an impoundment area of 27 hectares. Changes in habitat services were summarized in a habitat score combining variables such as riffle occurrence, sediment size, cover for fish and bank stability. Increases in the score were meant to reflect improvement in conditions for desirable fish species, in particular smallmouth bass. Following removal of the dam, the habitat score increased significantly in the former impoundment area and increased slightly in areas immediately below the former dam site and in free-flowing areas above the dam. Artificial creation of in-stream contours accelerated habitat improvements in some areas, but after several years habitat scores for areas undergoing only natural recovery had increased to similar levels. The study also noted an increase in the smallmouth bass population and a decline in the population of carp following dam removal.

Table 2.10. Increase in Habitat Services Five Years After Dam Removal

	Downstream	Upstream		
	0 to 0.8 miles	0 to 1 mile	1 to 1.5 miles	1.5 to 2.1 miles
Increase in Habitat Score as Percent of Maximum Score	15%	40%	55%	10%

Data from the Kanehl *et al.* (1997) study were adapted for use in scaling the Athos dam removal project. The relevant figures are reported in Table 2.10. Service increases were identified for four river segments, defined by distance from the former dam site. The first segment below the dam extended 0.8 miles downstream and was associated with a service increase of 15 percent. In the three upstream segments, services increased 40 percent, 55 percent and 10 percent, respectively. Based on Kanehl *et al.* (1997), it was assumed these service increases would be attained after a five-year linear recovery, starting at zero. The percentage service increases in a given year were multiplied by the area of aquatic habitat in each respective segment, and annual restoration gains were discounted at 3 percent. The delineated segments were applied to the Athos project without adjustments for any variation in the upstream extent of the impoundment area.

2.4.9.5 Creation of fish passage to compensate for aquatic habitat losses

Fish passage allows aquatic habitat upstream of a dam to be accessed by migratory species. It is reasonable to view this ecological change as an improvement in the integrity and value of the upstream habitat. The relative-habitat-value (RHV) approach to quantifying resource services is a useful way to evaluate benefits of this type of project. Steger and O'Connor (2001) and Iadanza (2001) introduced some of the relevant concepts, which depend on researcher consensus, and are often applied by mutual agreement between trustees and responsible parties. The example presented below is similar to applications in several recent cases.

Table 2.11. Increase in Habitat Services From Installation of a Fish Ladder

	Fish	Mammals	Birds	Total/Avg
Species-group weight	0.33	0.33	0.33	1.0
Aquatic habitat value				
Without restoration	0.6	0.55	0.65	0.6
With restoration	0.8	0.6	0.8	0.73
Fully functioning habitat	1	0.7	1	0.9
Increase as percent of full function				15%

Relative habitat values are usually analyzed with respect to representative species or species groups. The selected species are usually high on the food chain to account for ecological services throughout the food web. Cultural and economic values are also important criteria in selecting representative species and determining their relative weights for scaling purposes. Table 2.11 evaluates the effect of a fish ladder on upstream aquatic habitat with respect to fish, mammals and birds. Each species group is assigned a weight of 0.33 indicating equal importance across species in the scaling calculation.

The columns in Table 2.11 show the value of restoration to each species group. For example, upstream habitat for fish is assigned a value of 0.6 prior to installation of a fish ladder. With a fish ladder, upstream habitat is worth 0.8 with respect to fish. Both of these figures are estimated relative to the highest-value habitat for fish. In this example, a 1.0 in the final row indicates that fully functioning aquatic habitat has the highest-value habitat for fish. For mammals and birds, the estimated increase in services is based on the additional food that a fish ladder would provide to these taxa. The increase is 0.05 for mammals and 0.15 for birds, indicating that fish are considered a more important food source for birds than for mammals. The value of 0.7 for mammals in the bottom row reflects the judgment that aquatic habitat is not the highest-value habitat for mammals. This is because aquatic habitat is less important as a limiting factor for mammals than constraints on the availability of upland habitat. Sometimes quantitative metrics are available in the literature to support selected habitat values, but some site-specific judgments are usually required.

The last column of Table 2.11 shows the weighted-average habitat value with and without restoration. The weighting is based on the species-group weights. The increase in services (0.73 – 0.6 = 0.13) is divided by the value of fully functioning aquatic habitat (0.9) to obtain the estimated service increase of 15 percent. This figure would apply when scaling within aquatic habitat for both injury and restoration. Alternatively, cross-resource conversions could be implicitly incorporated into the scaling calculation by instead using the increase of 0.13 (or 13 percent). This lower figure implicitly weights the restoration of aquatic habitat according to the relative habitat value of 0.9, which may be higher or lower than the value of other habitats where injury or restoration occurs.

2.4.10 Enhancement of nesting success for turtles

Both freshwater and sea turtles have value to the public, have suffered significant declines, and are act risk to oil spills and other environmental incidents. The Endangered Species Act guarantees that any sea turtle injury be addressed with a population-level REA and several freshwater turtles will also be judged sufficiently important for that species-level of treatment in restoration scaling.

2.4.10.1 Creation of sandy beach nesting areas to compensate for turtle mortality

In the Chalk Point oil spill, an estimated 5,245 terrapin-years were lost from direct mortality of oiled diamondback terrapin adults and juveniles, from production foregone by their untimely demise, and from an estimated reduction of 10% in hatchlings in the succeeding breeding season from oiled substratum (NOAA *et al*., 2002). The restoration selected to compensate for this turtle loss was stabilization and enhancement of an eroding nesting beach. This project was paired with the marsh creation project, which provided a source of beach-grade sand from the excavation of farm fields to grade them down to suitable elevations for salt marsh grasses. That sand was to be used to supplement the narrow, eroding fringe beach and to grade it more gradually to the upland habitat behind, thereby replacing an abrupt erosion scarp. Future erosion was inhibited by installing two breakwaters off the beach rising to 0.6 m above the high tide water level to dissipate wind and storm wave energy before it reached the beach. In addition, the upland portion of this beach rehabilitation project was vegetated to prevent wind-blown losses and movement of the beach sands. Monitoring of diamondback terrapin nesting was planned so that if nesting use did not match the projections used in scaling restoration benefits to match the injury, then either nests or hatchlings would be moved to the rehabilitated beach to imprint this location so that females would return to utilize it for future nesting.

Scaling methods for this beach rehabilitation project involved application of known nesting densities of diamondback terrapins on beaches in this area, information on clutch size per nest and survival probability, and expected longevity of hatchlings from demographic data for terrapins (NOAA *et al*., 2002). Specifically, pre-project terrapin nesting density was assumed to be at the low end of natural nesting densities (240 nests per hectare) because of its steep eroding condition. After completion of the project, the nesting density was assumed to increase linearly from 20% of the average nesting density for diamondback terrapins of this area in the first breeding season after construction to 100% of the average after 5 years (683 nests per hectare), so the net benefit after full use is attained is 443 additional nests per hectare of rehabilitated beach. The average number of hatchlings produced per nest is 13 and the average nest survivorship in this area in 20%, meaning that by 5 years after project completion, a net augmentation of 443 x 13 x 20% or 1,150.5 hatchlings will be produced per hectare of rehabilitated beach. From knowledge of terrapin survivorship, each hatchling is expected to produce 2.095 discounted terrapin-years, meaning that 2410.3 additional discounted terrapin-years are produced per hectare of rehabilitated beach. The beach stabilization project was assumed to have a 25-year lifetime, from which the total discounted numbers of additional terrapin years produced was computed to be 34,233.4 per hectare, necessitating only 0.15 hectares (0.37 acres) of beach rehabilitation. The engineering constraints of the site actually required a minimum of 0.94 acres of beach, so the actual restoration contained nearly three times as much beach as scaling implied was needed for compensation. Nevertheless, monitoring of beach area persistence and terrapin nesting success was incorporated into the restoration so that any unanticipated shortfall in nesting could be corrected by importing nests or juveniles for imprinting to this newly augmented beach.

2.4.10.2 Mitigation of injuries to hatchling sea turtle by lighting

After sea turtles hatch, they rapidly march into the sea, using the lighted horizon to orient properly during the darkness of night. This orientation and thus their direction of movement can be disrupted by artificial lighting from streets, residences, and other beach-front development. Most sea turtle hatchings that fail to move directly to sea are killed by traffic, predators, or physiological stress. Where beaches are abundantly used for sea turtle nesting, especially south Florida, municipal ordinances exist to limit this light pollution and its negative impact on threatened and endangered sea turtles. However, the enforcement of such ordinances can be enhanced, thereby serving as a method of compensatory restoration for sea turtle injuries. In addition, replacing street lights that are elevated on poles with embedded lighting in median strips represents another more permanent method of providing mitigation for sea turtle hatchling losses. Both of these options, along with purchase and protection of nesting beaches, were considered as restoration projects to compensate for mortality of sea turtles during the Fort Lauderdale mystery spill (NOAA and FDEP, 2002).

The sea turtle injury from the Fort Lauderdale mystery spill included a few adult and older juvenile sea turtles but mostly affected recent hatchlings still near shore because of their higher density and greater susceptibility to the floating oil. The SIMAP (Spill Impact Model Analysis Package) model was used together with incident-specific information on estimated sea turtle abundance and susceptibility to estimate that 7,800 sea turtle hatchlings died plus 357 demographically equivalent hatchling equivalents from deaths of older turtles. This loss plus the interim loss of turtle-years between the time of injury and time of restoration constituted the quantitative loss that compensatory (and primary) restoration was scaled to provide for this Fort Lauderdale mystery spill.

Restoration scaling was achieved in units of hatchlings saved from death by disorientation from lighting visible on the beach. The project was directed towards two counties with appropriate beaches where mitigating lighting effects was judged to be potentially effective in saving hatchling sea turtles. In Palm Beach County, previous surveys and studies suggested an average of 9,191 nests exist annually, producing 80 hatchlings per nest and 3% natural disorientation rate. For the 19,610 nests expected annually in the enforcement area of Brevard County, these same estimates of hatchling production and disorientation rate were assumed. Thus summing across the enforcement area of the two counties implies that a total of 69,100 hatchlings are at risk from disorientation by beach-visible lighting. Enforcement was estimated to save 5% of these turtles annually or a total of 3,450 hatchlings annually, more than enough to match the estimated 3,002 hatchlings necessary for each of three years to replace the injury (discounted for the 5 years presumed between injury and initiation of restoration). This method of scaling restoration of sea turtle hatchlings by reducing beach light pollution depends upon availability of good estimates of the ongoing loss of hatchlings to disorientation from beach light pollution and the quantitative reduction achievable through enforcement of ordinances. For the alternative project of replacing street and municipal lights on poles with embedded lights in street medians, similar information would be needed on the quantitative effectiveness of the light reduction achievable.

2.4.11 Mitigation of human-use impacts

There is extensive and growing information about how human activities have inadvertent yet sometimes serious impacts on survival or reproduction of wildlife. The most widely reported of such impacts involves commercial fishing, where the term “by-catch” has been created to describe the capture (and usually mortality) of fish, birds, marine mammals, and other organisms that are unintentionally caught by nets, trawls, and other gear. Many of these by-catch mortalities in commercial fisheries have been quantified, leading to relatively confident quantitative scaling for compensatory mitigations or restorations that reduce by-catch. However, there are also many other human activities that lead to injuries to wildlife: these too provide extensive opportunities for use in compensatory restoration. However, collecting the quantitative information needed to compute scaling can be a challenge and a barrier to using the many creative concepts for restoration. Here, we do not attempt a comprehensive survey of such concepts, but instead merely discuss one example very briefly to illustrate the potential for this sort of restoration project.

2.4.11.1 Educational signage to reduce mortality of birds entangled in fishing lines

By running the SIMAP model of oil fate and the biological impacts sub-model, 12 seabirds, mostly cormorants, were estimated to have died through encountering sea-surface oil in the Fort Lauderdale Mystery spill (NOAA and FDEP, 2002). The project chosen for compensatory restoration was the erection of signs at recreational fishing piers near jetties in the south Florida spill area to educate the fishermen about how to free birds entangled by fishing lines instead of just cutting the lines. The signs provided a phone number to contact wildlife rescue personnel if needed. Scaling such public educational projects presents a challenge, but in this case previous experience with the public response to similar signs provided quantitative information on likely effectiveness. SOS (Save Our Seabirds) had placed 35 large signs at fishing piers presenting phone numbers of the organization instructing the public to call to get help in rescuing entangled birds. Records showed that 20 phone calls were recorded in a year’s time from those 35 signs providing a numerical rescue rate of 0.57 seabirds per large sign. Signs were estimated to have a 7-year life span. This quantitative information provided a basis for conservative scaling of the mystery spill restoration to compensate for seabird losses. Signs were erected at three locations on each of two popular fishing piers in the spill-affected region including instructions for freeing entangled birds and the phone number of the local Wildlife Care Center in nearby Fort Lauderdale. The birds saved by this restoration project are most likely to be pelicans and gulls, rather then the cormorants that suffered most from the spill, although two oiled pelicans were recovered during the spill. If a particular high-value species of seabird suffered injury, this type of project may not suffice to replace the losses in kind.

2.4.12 Mitigation of urban or agricultural run-off

Although much effort and cost in managing water quality has been directed toward improved treatment of wastewater before its discharge into streams, rivers, estuaries, and coastal oceans, the control of non-point source pollution from stormwater has lagged badly. Recent federal mandates from EPA for urban stormwater control and treatment may address this problem to some degree, but many opportunities for mitigation of uncontrolled (sub)urban and agricultural stormwater run-off and pollution. Here we barely ripple the surface of the pool of possibilities by suggesting two interventions as potential compensatory restorations.

2.4.12.1 Collection and treatment of urban run-off to compensate for oil release into marine waters

Because stormwater run-off from impervious urban and suburban pavements like roads and parking lots provides a direct injection of PAHs in various stages of weathering into the receiving waters, controlling this process ranks high as one means of direct mitigation for oil spilled into those same waters. This has particular appeal and likely applicability to urban estuaries, which receive so much oil-derived pollution in stormwater. In the Julie N case (MDEP *et al.*, 2000), compensatory restoration for marine community injuries in the Portland Harbor area were to be restored by purchase of a vaccum truck by the City of Portland to allow cleaning of oil- and grease-contaminated sediments from the city's stormwater collection system and thus preventing their discharge into the estuarine harbor waters. This mitigation for an ongoing source of petroleum pollution was designed to compensate for injuries to the marine community, documented explicitly as oiling of blue mussels, soft-shell clams, and estuarine sediments, plus oiling and washing injuries to epifaunal communities on bulkheads and shorelines. While this approach has merit, in the Julie N case there was no quantification of injury in terms that could lead to confidence in achieving full compensation, a potential drawback of this approach.

2.4.12.2 Reduction of agricultural run-off to compensate for spill-related nutrient loading

We have characterized the process of scaling to compensate for natural resource and habitat injuries as one in which we identify a metric for ecosystem services, either at the resource (population) level or at the habitat scale and use that metric to size the necessary restoration project in a REA or HEA. However, some injuries may result from stressors that are fundamentally different from the oil and other toxic chemicals or from the physical habitat damage that underlie the damage assessment and restoration scaling processes that we have described to this point. In such cases, the guiding principle for scaling restoration or mitigation should be application of an intervention that matches and equals in the opposite direction inputs or effects of the stressor. The spill of acidic process water into the Alafia River in the Mulberry Phosphate case (Mulberry Phosphate Trustee Council, 1999a,b) represented not only a toxicant to fish but also the loading of a plant nutrient into the river and estuary. Because this aquatic system like most others is already stressed by excess nutrient loading, which leads to loss of ecosystem services from eutrophication, trustees for Mulberry Phosphate developed a REA model using a currency of nitrogen loading. The preferred restoration action was improvement of riparian habitat along the river and estuary margin to produce nutrient, especially nitrogen, uptake in amounts sufficient to mitigate for the nutrient content of the spill. Because riparian buffer preservation and establishment represent a standard BMP for water quality maintenance, there are available data on which to estimate the quantitative benefits of buffer establishment and other BMP improvements so as to conduct scaling to match the spilled load.

2.4.13 Reduction in commercial harvest

Injuries to exploited fish and shellfish stocks can be restored through compensatory restoration projects that act by reducing commercial harvest to match the size of the injury. This action operates by leaving alive in the ecosystem those animals that would otherwise have predictably been extracted by fishing. Use of this REA method of compensating for injury to an exploited resource has relatively low uncertainty associated with it because fishery statistics like stock assessments and landings are so commonly taken that success can be readily quantified. On the other hand, implementation would require some assurance that the reduction in harvest in one or more years would not be negated by increased fishing pressure in subsequent years and consequently enhanced harvests driven by fishermen’s perception and expectation of abundance. Implementation of this method of compensatory restoration must also achieve a means of ensuring that the compensation for the loss of a public resource is not achieved by inflicting a private injury on fishermen and the fishing industry by preventing commercial or recreational fishing without compensation.

Compensatory restoration of lobster injuries from the North Cape oil spill serves as a model example of how this approach of reducing commercial harvest might be scaled and implemented in future applications (French McCay *et al*., 2003a). The injury to lobsters from the oil spill had been quantified by counting dead lobsters stranded on shore by sex and size (convertible to age) and by systematically sampling lobster density on the seafloor in- and outside the spill area. Then an entire demographic life table for American lobsters was formed from known longevity, age-specific mortality, and age-specific female fecundity information. Assuming that the lobster population was in equilibrium, with each lobster exactly replacing itself, the fecundity data were used to infer survivorship of eggs to the earliest benthic life stage, forming a complete life table. This allowed conversion of numbers of lobsters of any age to demographically equivalent numbers at any other age. Hence, the entire kill was converted into the demographically equivalent numbers of eggs required to produce the lobsters killed. Because lobsters are so heavily fished, it is reasonable to assume that the population is in part limited by reproduction so that a method of providing additional egg production equal to what is required to replace the lobsters lost would represent compensation for spill mortality.

Scaling a reduction in fishing pressure to achieve compensatory restoration of lobster injury could be done in several alternative ways, ones that use egg production as the metric and others that use lobster production as the metric (French McCay *et al*., 2003a). For valuable, managed fish stocks like American lobsters, fishery management employs modeling of "surplus production", which is the estimated amount that can be sustainably harvested, that express that production as a function of fishing mortality rate (F), natural mortality rate, and stock biomass (e.g., Ricker, 1975; Prager, 1994). Application of this fishery modeling approach to scaling injury and matching it to restoration would employ the metric of production foregone because of the acute mortality and harvest production foregone by reducing F. The production foregone because of injury would add the biomass of those individuals that were killed to the biomass that they would have been expected to produce before death in the absence of the spill. Such a method is widely applicable to fished species whose harvest is already managed by surplus production modeling.

Alternative means of scaling reduction in fishing pressure to restore lobster injury utilize the metric of augmentation of numbers of lobster eggs required to replace all losses of lobsters. Use of this metric can be achieved by several alternative means of manipulating fishing impacts. Because adult lobsters can be marked in a clearly recognizable way by v-notching their carapace at the tail and this mark lasts through 1-2 molts, fishing mortality can be temporarily reduced by applying and enforcing regulations that prevent possession, sale, or use of any v-notched lobster. The lobster restoration method that was chosen for the North Cape oil spill used this ability to mark lobsters to design a restoration project that involved purchase of adult female lobsters from fishermen, v-notching them, and freeing them under no possession regulations. This intervention has the value of increasing egg production over that would have existed under the fishery by allowing those females to produce additional eggs that would otherwise not have been. The advantage of this method of reducing fishing mortality is that fishermen are not penalized by suffering harvest restrictions in the process of reducing fishing mortality. The method is not as broadly applicable as reducing fish harvest by further limiting the allowable catch, and some means of compensating fishermen for the profit reduction of limiting their allowable catch should be viable for finfish to achieve this same goal of avoiding compounding the injury.

Quantitative scaling to match the injury based on numbers of additional eggs produced by the females whose v-notching allows them to survive longer is done by relatively straight-forward application of demography using mortality and fecundity data summarized in a life table (French McCay *et al*., 2003a). The notched females have higher survivorship than unmarked ones, and life table analyses compute how much additional egg production this achieves. For the North Cape example, the size (age)-specific losses were known by sex from the injury assessment. Mortality rates and the assumption of steady state population under normal harvest conditions allow these observed mortalities to be converted into the numbers of eggs required to have produced them. Accounting for females only, a standard convention in life table analysis, which assumes that male numbers will follow, the numbers of new eggs produced to restore the observed mortality was 11 billion, discounted for time lags both between date of injury and date of v-notching and also between egg production and time required to grow into the size (age) of the lobsters killed. The computation of numbers of new eggs required for quantitative restoration assumed: (1) a 5% handling mortality in the notching process from past experience; (2) fishing mortality would still exist but at 50% of the normal rate through the first molt and

75% of normal after the second molt, after which no protection was afforded; and (3) other demographic parameters remained unaltered by the return of notched females (French McCay *et al.*, 2003a). Thus this modified life table could be used to compute how many additional eggs are expected per 90-mm notched female (the average size in the harvest) during her remaining expected lifetime, which amounted to 4260. After applying discounting for all time lags, the numbers of v-notched lobsters required to compensate for the injury was estimated to equal about 3.6 million. To avoid incurring limitations in food, rocky reef habitat, or density-dependent predation or cannibalism, this number of v-notched lobsters was spread out over a 5-year period, a generally wise procedure when the restoration is large enough to risk temporarily unbalancing the ecosystem.

2.4.14 Marine resource stocking programs

Technology exists to produce many shellfish and finfish species in culture operations in hatcheries and nurseries, suitable for transferring into nature to compensate for injuries to those same or analogous species. Among finfish, salmonids, trout, striped bass, and red drum represent examples of valuable species that can be cultured. Bivalve mollusks are the guild of shellfish for which effective hatchery/nursery programs exist that would enable artificial propagation to be employed as a restoration process. Beyond commercially and recreationally exploited aquatic species, there has also been successful development of culture programs for sea turtles and much ongoing work to propagate endangered species of a wide variety of taxa. These all have potential for use in compensating for damages caused by environmental incidents, although careful consideration is needed of the plusses and minuses on a case-by-case basis.

2.4.14.1 Augmentation of shellfish stocking programs to compensate for shellfish mortality

The North Cape oil spill off Point Judith resulted in mortality of many species of bivalve shellfish, essentially all of which can be grown in hatcheries for transfer and augmentation of natural populations. Losses occurred to surf clams, quahogs, soft-shell clams, American oysters, razor clams, and bay scallops. Shellfish hatcheries exist with the technical capacity to spawn and culture each of these species. This was a restoration option considered carefully by the North Cape trustees. Scaling for this REA is very straight-forward in principle. Because information exists on age-specific survival and growth, a life table can be constructed (French McCay *et al.,* 2003b) and both the injuries and mitigation can be expressed in the form of numbers of shellfish in equivalent life stages (ages and sizes), facilitating quantitative compensation.

Despite these obvious advantages to using culturing of seed shellfish as compensation for injuries to the very same species in the very same locations, North Cape trustees chose not to employ this option except in a limited way for bay scallops. Given the limited number of qualified shellfish hatcheries and their limited capacity, restoration could be achieved only over some period of years to match the scale of the injury. This delay was considered a negative aspect of the culturing option. Costs were higher than some alternatives including the one selected, which involved purchase of quahogs from fishermen and stocking them in a salt pond closed to fishing so that the biomass lost from deaths in several species was then replaced by growth of larger clams of one species. Bivalves lost to the oiling were largely small, almost all smaller than those used for the restoration. This form of augmenting production does provide some of the ecosystem services of suspension-feeding bivalves, especially water filtration. However, by excluding fishing, the clams do not enter the fishery and by achieving production only of larger clams, the food chain services of smaller clams to crabs, demersal fishes, and some birds are not provided because it is the smaller quahogs that are consumed. Furthermore, packaging restoration of an entire guild of injured suspension-feeding bivalves into a single species for restoration purposes would appear to contradict the resource agency commitments to sustaining biodiversity. The rapid introduction of large numbers of this single species also creates an unnatural community composition.

2.5 Habitat and resource conversions

Habitat and resource conversions can be useful in a variety of circumstances. For example, losses to river sediments cannot be offset by creation of new river bottom, so conversion factors allow marsh or oyster reef creation to substitute for the services of river sediments. As we discussed earlier, low-value habitats are not readily restored and would provide inefficient returns for the costs of restoration. Often it is efficient to use one type of restoration to offset losses to multiple species, so losses to both fish and birds are converted to an equivalent marsh-restoration metric. In some cases acute, compensable impacts occur to one habitat type, but due to long-term degradation or scarcity, another habitat type is the preferred focus of regional restoration efforts. High-value habitats are also those most seriously depleted and in most need of restoration, thereby motivating trustees to choose salt marsh, oyster reef, mangrove forest, coral reef, or seagrass for habitat restoration even if unvegetated sedimentary bottom were the habitat suffering injury. The ability to convert between habitats satisfies the dual objectives of natural resource compensation and long-term ecosystem management.

There are two basic approaches to habitat and resource conversions. One uses energy (biomass production) as a metric, with conversions across habitats or resources evaluated at a common trophic level (French McCay *et al*., 2003b; Peterson *et al*., 2009). If productivity at one or more critical trophic levels in a *Spartina* salt marsh (measured in grams of production per square meter per year) is greater than productivity in a seagrass bed, then an acre of seagrass would convert to less than one acre of salt marsh. The relationship between energy and value is primarily assumed based on the professional judgment of scientists, though some support for this approach is available in the ecological economics literature. For example, Costanza *et al*. (1997) observe a direct relationship between net primary productivity and the value of ecosystem services for a variety of habitats in various locations, and Ingraham and Foster (2008) use this relationship to predict differences in ecosystem value across geographic regions.

The second approach to conversions directly relies more directly on the professional judgment of biologists and ecologists. The metric is value, as determined by scientists examining the services provided by alternative resources and habitats (Stratus Consulting, 2000; Iadanza, 2001; TGLO *et al*., 2001; Bailey Trustee Council, 2003). The advantage of evaluating services directly is that multiple services can be considered jointly, and conversions do not rely on the assumption that one service, such as production, is representative of all other services. While many practitioners are hesitant to rely on this method because it appears subjective, the perception that other methods are less subjective is misleading, as discussed further in Section 2.7 on economic principles in compensatory restoration.

2.5.1 Animal biomass conversions

Once injury assessment is complete and a metric has been established on which a REA or HEA can be conducted to ensure that the restoration compensates for the magnitude of the injury, trustees face a choice of what restoration project to pursue. Ordinarily, replacement of lost resources in kind is the preferred option, as is locating the restoration within the same local geographic area where the injury had occurred. If an injured resource is highly valued as a species because it is federally listed as Threatened or Endangered under the Endangered Species Act, on a federal or state list of species of concern, is a marine mammal under protection of the Marine Mammal Protection Act, is exploited as the target of a valued fishery or as game, or achieves a special status as an iconic species for the region, then replacement in kind is almost certainly required. Fortunately for the trustees, these especially threatened and valued species usually are the subject of a federal Species Recovery Plan or a management plan that provides not only in-depth biological information about the stock but also identifies numerous potential restoration or management actions that could serve to enhance or protect the population. This makes the trustees' job easier because these plans have already received critical evaluation and considerable thought by experts on the species. The results of the planning process thus represent a menu of thoughtful choices among which the trustees may choose. For example, the restoration of piping plover injury after the North Cape oil spill involved nesting area protections on the limited sand beach habitat, for which past experience with implementation elsewhere also allowed confident scaling of the success (Donlan *et al*., 2003). The existence of a Piping Plover Recovery Plan into which the U.S. Fish and Wildlife Service and many others had put great effort thus provided professional expert guidance, results of past management interventions, and thus enhanced the confidence of success in the compensatory restoration alternative that was selected.

In many injury cases, natural resource damage assessments yield relatively long lists of species resources that suffered injury and provide quantitative assessments of injuries for each of them separately or as guilds of similar resources. For many reasons, trustees rarely consider establishing separate species-specific restorations to compensate for each these losses in services of multiple similar resources. The costs of preparation, evaluation, and conducting several independent restorations would be high and could readily exceed the value of the injury. For many species or resources, insufficient knowledge may exist to allow trustees to determine the key factors limiting the respective populations. Absent such knowledge, it is difficult or impossible to craft a species-specific intervention that would provide confidence in its success. For many species or resources, technological details and procedures for successful culture or husbandry may be lacking. Under these conditions, restoration plans are typically developed and implemented to handle large groups of similar species. For example, the North Cape oil spill caused mortality and thus service losses in several species of benthic bivalve mollusks, recognized of importance in the ecosystem as filters of the water and serving to support recreational and commercial fisheries. These included surf clams, oysters, quahogs, bay scallops, and soft-shell clams (French McCay *et al*., 2003b). Restoration planning contemplated and eventually selected restoration of a single species of these bivalves, the quahogs, to replace the lost ecosystem services of them all. This decision represented a cost-effective choice and allowed selection of the most reliable option for replacing lost bivalve production and water filtration services. On the other hand, propagation techniques are available for each of these species, so no technological limitation exists to replacing them in kind. By doing so, trustees would have retained the biodiversity of this guild rather than letting one serve the role of many (Peterson and Lipcius, 2003). Maintaining biodiversity for its value in supporting ecosystem resilience to various natural and anthropogenic stressors is a key strategy in conservation (e.g., Naeem and Li, 1997). These species do not even occupy the same habitats, with surf clams in sands off the ocean beach and the others in salt ponds, bays, and estuaries. Consequently, one might conclude that cost issues overrode conservation and restoration principles in this example, and are likely to do so again under present administrative processes.

A more common restoration approach to compensate for guilds of injured species is to conduct a habitat restoration or protection project. Such projects benefit a wide range or resources and thereby help sustain biodiversity while compensating for total service losses of the injured species as scaled to a common metric. The metric of choice for scaling such group restorations is biological production at the relevant trophic level. For example, the spill of acidic process water from a phosphate facility into Alafia River off Tampa Bay caused a large fish kill of numerous species, for which compensation was achieved by scaling fish gains from restoring oyster reef habitat (Peterson *et al*., 2003b). Thus, this example represents scaling of a guild of ecological similar animals feeding largely at the third trophic level (primary predators).

Injury assessment parameters must be chosen in contemplation of the type of restoration. If a species resource is to be replaced in kind with a species-specific restoration, then the injury and benefits are commonly scaled in units of organism-years. One organism-year for a given species is the presence of one individual for a year. This metric relates directly to population counts over time, which is responsive to the importance of conservation at the species level for valued species resources. If a guild of species is to be restored as a group, then the appropriate metric is usually the production lost at the trophic level represented by the guild of organisms suffering injury. This includes the biomass of organisms suffering acute mortality, the biomass production foregone by preventing those dead organisms from living out the rest of their expected natural life span, the biomass lost by sub-lethal reductions in growth of survivors, and the biomass lost by any reduced production of subsequent generations if multiple-generation impacts occur. Inferring the potential multi-generational impacts is one of the most difficult challenges facing the trustees in computing injury and scaling restoration because consequences of changing reproductive stock on recruitment of subsequent generations are widely debated in basic ecological literature.

When injuries occur across multiple trophic levels and a habitat restoration project is proposed for compensation, a question arises over if and how to combine production losses across trophic levels. This concern arises over whether combining injuries across different trophic levels represents double counting and results in overcompensation if restoration scaling were based on combined losses. If combination of losses across trophic levels is justified, then the methodology of combining represents a challenge. These issues are addressed and answered in a scaling exercise done by French McCay and Rowe (2003) done to scale compensatory restoration for injuries from the North Cape oil spill to suites of benthic invertebrates, fishes, and aquatic birds. The benthic invertebrates occupy the second trophic level (herbivores and detritivores), whereas the fish and birds are largely primary predators consuming these invertebrates. French McCay and Rowe (2003) argue compellingly that the injuries documented to multiple trophic levels are indeed additive. The argument turns on the recognition that although higher trophic levels are indeed derived by consuming lower trophic level production, the lost animals at higher trophic levels, such as primary predators, had consumed other secondary producers, not the ones that injury assessment documented as losses. Consequently, the trophic level losses in biomass killed plus production foregone are independent and additive.

The process of addition, however, is not simple summing because the units of primary, secondary, and tertiary production are intrinsically different. The classic Eltonian assumption is that energetic conversions from one trophic level to the next occur at an efficiency of 10% (Lindemann, 1942). Under that efficiency, 1 kg of lost primary predator production is equivalent to 10 kg of lost herbivore production, for example. French McCay and Rowe (2003) used emerging knowledge of ecological efficiencies of energy transfer from one trophic level to the next in the relevant estuarine habitats suitable for compensatory restoration in New England, salt marsh and seagrass, to model ecological efficiencies and thereby compute the transformations that would express production at the second and third trophic levels to equivalent units of the amount of primary production required to produce that animal production. Once expressed in this common unit of primary production equivalents, then total injury and total gain from restoration can be computed by summing these equivalents across all trophic levels. This set of empirically based computations provides a meaningful method of dealing with resource equivalency analyses that can do such scalings as required to combine losses across multiple trophic levels. The assessment of ecological efficiencies of trophic-level transfers used in French McCay and Rowe (2003) is an area of ongoing research and updated efficiencies appear in Peterson *et al.* (2009).

2.5.2 Habitat conversions

The simplest form of compensatory restoration for the services provided by an injured habitat is to conduct restoration of that very same type of habitat. In that way, lost services are replaced in kind. For many reasons, restoration of a different habitat may be preferred. For example, injury to either intertidal or subtidal unvegetated estuarine soft-bottom benthic habitat would be difficult to replace by restoring either of these habitats. By definition, neither contains substantial biogenic structure, so addition of structure is not a viable restoration technique. Grading of terrestrial lands to reclaim estuarine bottom seems unlikely to be feasible in that waterfront property is so expensive and if the shoreline possesses a wetland margin, the entire wetland complex would need to be destroyed and moved inland to make room for additional intertidal or subtidal unvegetated bottom. Beyond the issue of feasibility, there is an issue of spatial scope of the project. Unvegetated intertidal and subtidal benthic habitat is considered the least productive of estuarine shoreline habitats, so if injuries are compensated by restoration of an alternative, more productive habitat like a slat marsh, then the area required for the restoration is much smaller. Finally, the more productive habitats like tidal marsh, seagrass bed, oyster reef, and mangrove forest have been dramatically destroyed and degraded, so their restoration is often a management priority. Depending on which of these highly valued habitats is most depleted, the injury to one of these valued habitats may be most appropriately mitigated by restoration of one of the other valued habitats. The reliability of restoration techniques may vary among habitat types, with past experience providing greater confidence in the capacity to perform one type of restoration more effectively than other. Finally, opportunities for restoration may vary by habitat type in a system experiencing some damage from a spill or mechanical injuries to a habitat. One of the best indicators of potential restoration success is evidence of previous occupation of a site by the habitat to be restored. Such "orphan" sites may exist for one valued habitat but not for the one injured, providing motivation for habitat conversion during compensatory restoration.

When a habitat conversion is chosen, clearly a metric of ecosystem services must be employed that can be compared across habitats. The question that needs to be answered is how valuable a unit area of one habitat type is compared to the one that is being replaced, as indicated by an acceptable metric for total ecosystem services. This question has typically been answered by a formal process of obtaining best professional judgment from a group of informed experts. For example, in the Lavaca Bay case (TGLO *et al.*, 2001) injury to subtidal bay bottom was compensated by salt marsh restoration. Salt marsh habitat was chosen over oyster reef habitat largely because the types of resources produced by marshes were considered more similar to those suffering injury in the subtidal benthic habitat. To determine the appropriate habitat conversion ratio for replacing lost acre-years of subtidal bottom services with salt marsh services, a group of local Texas experts was asked to provide a quantitative ratio of per-acre services based on 5 criteria: (1) primary production; (2) secondary production; (3) benefits to fish and decapod crustaceans; (4) detritus production; and (5) and remineralization. The outcome of this process rated ecosystem services from an acre of salt marsh 5 times as valuable as an acre of subtidal bottom, so this was the conversion factor applied to scale the restoration project.

Similar judgments were made by experts in the Bailey Superfund case in Texas, the Green Bay Superfund case in Wisconsin and Michigan, and the Commencement Bay case in Washington. In the Bailey case (Bailey Trustee Council, 2003) a group of 6 experts (three representing the responsible parties and three government scientists) determined how to relate ecosystems services of sub-habitats within the marsh complex to a common standard of brackish tidal marsh to which all injuries were to be converted by restoration. The results of this "multiple attribute decomposition" process were as follows: brackish tidal marsh – 1.0; high marsh – 0.61; freshwater marsh – 0.79; marsh ponds – 0.56; ditch habitat – 0.43; and uplands – 0.41. Details of the expert ratings were provided previously in Table 2.5. In the Green Bay case (Stratus Consulting, 2000) experts for the natural resource trustees developed several habitat conversions for evaluating restoration options. A ratio of 3:1 was used to evaluate preserved wetland relative to restored wetland, indicating a preference for higher-valued preserved wetland. A ratio of 2:1 applied to coastal and other "high-quality" wetland relative to wetland in more populated areas. A ratio of 9:1 applied to coastal wetland and coastal upland habitat. In the Commencement Bay case (Iadanza, 2001), the following habitat conversions were developed: estuarine marsh, 1.0; intertidal, 0.75; shallow subtidal, 0.55; deep subtidal, 0.3; rip-rap, 0.1.

Other processes used to determine habitat conversion ratios to allow compensatory restoration of a habitat different from the one that was injured have been based on more objective quantification of a service, productivity. The Chalk Point oil spill (NOAA *et al.*, 2002) produced a net loss of production of benthic invertebrates (declines in two bivalves and an amphipod, partially mitigated by an increase in polychaetes), fish and mobile shellfish, and birds (excluding ruddy ducks for which a separate restoration project was implemented), all of which were compensated by an oyster reef restoration. The metric used to conduct this scaling conversion between augmented oyster and associated mud crab, grass shrimp, and small crustacean production and lost production of benthic invertebrates, fish and mobile shellfish, and birds was production at the trophic level on which the target species or guild feeds. So the benthic invertebrates were replaced by other benthic invertebrates so no efficiency of conversion to a higher trophic level was applied to match the necessary 2,256 kg of their lost production. For fish and mobile predatory shellfish, a 20% efficiency of conversion from oysters and associated invertebrates was assumed in computing the replacement for 2,464 kg of lost production. To replace the birds estimated to have died, the dead bird numbers were converted to biomass of birds. For birds that eat benthic invertebrates an ecological transfer efficiency of 2% was assumed and for those that eat fish a 0.4% efficiency was assumed. The oyster reef area required to produce the sum of the production required to match these losses in three different guilds totaled 4.69 acres. The benthic invertebrate production benefits associated with oyster reef restoration was empirically determined by review of locally relevant literature on oyster growth and survival of seed oysters stocked onto the reef and augmented abundances of oyster reef associates.

An analogous procedure was conducted to convert ecosystem service losses of filling water column and burying unvegetated sedimentary benthic communities during expansion of the Craney Island port in the Chesapeake Bay into oyster reef and salt marsh services (Peterson, 2003). Production at the second trophic level (herbivores) was used as the metric on which ecosystem service losses were scaled to match gains from habitat restoration. This then represents replacement of services of the water column habitat and the unvegetated benthic habitat with services from oyster reef and salt marsh habitat. As opposed to the Chalk Point case, where invertebrate production losses where directly estimated by seasonal change in biomass of invertebrates on oiled bottom, for Craney Island, the lost production was estimated from compiling local data on macrozooplankton biomass in the water column and benthic macroinvertebrates on the bottom and converting biomass (B) to production (P) by multiplying by P/B ratios from the literature. To maintain an analogous approach in computing credit from restoration, literature reviews of local data on macrobenthic biomass on oyster reefs enabled estimation of augmented macrobenthic production associated with constructed oyster reefs. For salt marshes, insufficient macrobenthic biomass data were available so scaling was done on the basis of converting primary production to secondary for comparison to the losses in the two affected habitats.

The most recent development in the methodology for conversions among habitats uses multiple ecosystem services, as considered in the best professional judgment approaches, but done using quantitative data from syntheses of all available literature. Peterson *et al*. (2009) collected all published and unpublished data from estuaries from Texas to Massachusetts on production and/or biomass of primary, secondary, and tertiary producers in the 5 common shoreline habitats: salt marsh, unvegetated intertidal flat, oyster reef, seagrass bed, and subtidal unvegetated flat. From this data synthesis, the average primary, secondary, and tertiary production was computed in each habitat. This synthesis then allows productivity comparisons among habitats at each trophic level and contrasts among habitats based on comparing sums of production equivalents across all three trophic levels. Peterson *et al*. (2009) computed production equivalents in two different ways: one using classic Lindemann (1942) expectations of 10% conversion efficiency between adjacent trophic level and a second using ecological efficiencies estimated empirically from this review of production at each trophic level. Because vascular plant production of salt marsh and seagrass habitats largely enters detrital food chains in which conversion to microbes precedes assimilation by herbivores (detritivores) and this occurs at a cost of energy loss, Peterson *et al*. (2009) used present understanding of estuarine food web processes to convert primary production into production suitable for consumption by herbivores (all microalgae) plus production suitable for consumption for detritivores (the bacteria and fungi decomposing vascular plant detritus). Empirical conversion efficiencies from the first trophic level to the second were 26% and from the second to the third 27%.

Table 2.12. Ratios of productivity comparing pairs of estuarine shoreline habitats using average production data from Peterson *et al*. (2009).

Habitat Comparison	Trophic level 1°	2°	3°	Average across 3 trophic levels	Based on 10% weighted sum	Based on 26-27% weighted sum
(1)Oyster reef to:						
Intertidal flat	1.5	21.8	2.1	8.5	8.4	10.5
Subtidal flat	2.4	22.4	2.0	8.9	8.5	12.1
Spartina marsh	0.5	24.8	2.0	9.1	7.0	6.3
Seagrass bed[1]	0.3, 0.2	20.7, 18.4	3.1, 7.8	8.0, 8.8	7.6, 7.3	5.1, 3.8
(2)*Spartina* marsh to:						
Intertidal flat	3.1	0.9	1.1	1.7	1.2	1.7
Subtidal flat	5.1	0.9	1.0	2.3	1.2	1.9
Seagrass bed[1]	0.7, 0.5	0.8, 0.7	1.6, 4.0	1.0, 1.7	1.2, 1.3	0.8, 0.6
(3) Seagrass bed[1] to:						
Intertidal flat	4.3, 6.5	1.1, 1.2	0.7, 0.3	2.0, 2.7	1.0,.0.9	2.1, 2.7
Subtidal flat	7.1, 10.6	1.1, 1.2	0.7, 0.3	3.0, 4.0	1.0, 0.9	2.4, 3.1

[1] Computations presented separately for *Zostera marina* and *Thalassia testudinum* respectively.
[2] Weighting was done by energetic equivalents, using Lindemann 10% ecological efficiencies to convert production at each trophic level to the 1° production of usable foods required to produce that 2° or 3° production.
[3] Weighting was done by energetic equivalents, using inferred ecological efficiencies of 26%-27% to convert production at 1° to 2° and 2° to 3° levels respectively.

The documentation of primary, secondary, and tertiary productivity in each estuarine shoreline habitat can be used in multiple alternative ways to compute habitat conversion ratios (Table 2.12). Under several alternative procedures, three different ecosystem services, production at each of three trophic levels, are quantitatively combined to form habitat conversion ratios. First, one could simply average the between-habitat productivity ratios to compare any given pair of habitats. Second, one could use the traditional 10% ecological efficiencies to compute the sum of primary production equivalents for each habitat and construct between-habitat ratios of those sums. Third, one could use the empirically derived 26 and 27% efficiencies and recomputed the sums of primary production equivalents and recalculate the between-habitat ratios. Independent of method, oyster reef habitat exceeds all others in total productivity across the lowest three trophic levels: 8-9 times (using averages), 7-8 times (using Lindemann efficiencies) and 4-12 times (using inferred efficiencies) other estuarine shoreline habitats (Table 2.12). *Spartina* marsh exhibited higher average productivity ratios and total production using inferred transfer efficiencies than intertidal and shallow subtidal flats by factors of about 2, although the corresponding ratio of sums of primary production equivalents was only 1.2 assuming Lindemann efficiencies. Seagrass beds also exhibited higher tri-trophic productivity than both tidal flat habitats by average productivity ratios of 2-4 and ratios of total production of 2.5-3 using inferred efficiencies; again assuming Lindemann efficiencies of 10% made the difference between seagrass and intertidal and subtidal flat habitats in total tri-trophic production essentially disappear. Contrasts of *Spartina* marsh to seagrass habitat productivity are ambiguous. Averaged across all three trophic levels, marsh to seagrass productivity ratios were slightly greater than unity at 1-1.7, while the corresponding ratios of total production under Lindemann efficiencies were 1.2-1.3. However, using inferred ecological efficiencies, however, seagrass habitat appeared more productive than *Spartina* marsh by a factor of 1.4 (Table 2.12).

2.6 Theoretical support and limitations in the HEA/REA model

Restoration scaling is based on the premise that gains must offset losses. The discussion up to now has not addressed the issue of precisely what this means. Does every affected individual receive benefits that fully offset her losses, or do total gains offset total losses for the affected population? If the goal is to offset total losses, how are individual values added up? Often in economic theory, money is the common denominator that allows individual preferences to be combined into a measure of collective value. In HEA/REA applications monetary value is removed from the scaling calculations. This creates a potential gap between theory and methods. To maintain consistency with a meaningful standard for determining appropriate compensation, HEA/REA practitioners must accept certain assumptions regarding what people want and what compensation they are entitled to.

If everyone in the affected population is impacted in the same way and equally by both injury and restoration, the measure of appropriate compensation is simple. When the scale of restoration is large enough to satisfy any given person, it is large enough to satisfy everyone. This is the simplified scenario that underlies the early development of HEA. In Unsworth and Bishop (1994), injury to marsh was offset by restoration of marsh and the injured and restored marshes were assumed to be perfectly interchangeable. The possibility that different people could have different preferences for specific characteristics of injured and restored services was not addressed. For example, consider a case where losses occur from contamination of a marsh and restoration is undertaken to create marsh habitat in an uncontaminated area of the same watershed. Opinions could vary considerably regarding the harm associated with contamination in the environment or the appropriateness of habitat creation. Someone who places a high value on a clean environment and a low value on manmade marsh is not likely to be fully compensated if HEA is performed based on average preferences in the population.

Economic theory offers a way to examine collective value when individual values differ. In particular, benefit-cost analysis relies on the premise that a project or policy is acceptable from a public-choice perspective if the sum of individual values for the project or policy is positive. Individual value for those who benefit is measured as the amount they would be willing to pay to undertake the policy. For those who do not benefit, the value may be zero, or it may be the amount they would be willing to pay to prevent implementation of the policy. The unfair treatment of those who lose from a particular action is justified by the notion that "winners" can make payments to compensate "losers" up to the point where everyone is better off. This approach to collective decision-making is known as the Kaldor-Hicks criterion, after the economists who first developed it.

Jones and Pease (1997) suggested that the Kaldor-Hicks criterion could be applied to compensatory restoration. If those who are overcompensated by restoration obtain enough surplus value to make up for those who are not fully compensated, then the total level of compensation could be viewed as adequate. This provided the necessary theoretical foundation for a shift away from the use of lost value (or "compensable value") as the basis of a damage claim (DOI, 1986) toward an approach that calculates damages based on the cost of appropriate restoration projects (NOAA, 1996). Jones and Pease also presented a slightly expanded version of the HEA/REA equation, allowing some variation in individual preferences. However, they retained the critical simplifying assumption of Unsworth and Bishop (1994) that a single level of resource or habitat restoration simultaneously satisfies the preferences of all individuals. The gap between theory and methods in HEA/REA was left unresolved, because the Kaldor-Hicks approach to collective compensation is never invoked when compensation is simply assumed to satisfy everyone.

Flores and Thacher (2002) further examined the issue, and showed that metric-based models of compensatory restoration are generally not compatible with economic principles of individual compensation. Specifically, they pointed out that differences across individuals in the amount of compensatory restoration that they require prevent the cancellation of monetary value from the HEA/REA formula. This contradicts the standard assumption of the HEA/REA model, in which cancellation of monetary value allows calculations to proceed using only biological parameters. Thus the HEA/REA model assumes identical preferences at the outset, and the issue of collective versus individual compensation never arises. Flores and Thacher further observed that attempts to address diverse preferences using metric-based analysis, for example by assuming that biological metrics represent an average of individual-specific restoration requirements, will generally fail to satisfy criteria for a theoretically valid measure of collective compensation. In the final analysis, the simple formulation presented in Unsworth and Bishop (1994) may represent the conceptual limit of the HEA/REA model in defining appropriate compensation.

HEA/REA models rely on a variety of simplifying assumptions due to both theoretical and practical limitations. Several researchers have investigated the effect of these assumptions on the results of scaling calculations. Flores and Thacher (2002) went on to show that certain structures for the distribution of individual preferences can be represented by average preferences without altering scaling calculations. Average values would then cancel out in the comparison of losses and gains, and HEA/REA results would reproduce the outcome of a more complete Kaldor-Hicks compensation scheme. While the necessary preference restrictions are limiting, they are also common in many econometric models that value resource services using linear, representative-agent specifications (e.g., Adamowicz *et al*., 1998a; Hanley *et al*., 1998). Dunford *et al.* (2004) examined the sensitivity of HEA/REA to a host of biological and economic assumptions and found considerable sensitivity in scaling outcomes. For example, they examined the assumption that the marginal value of resources remains constant through time and across various levels of resource abundance. They found that required restoration varied by as much as 66 percent when a two-percent annual change in the marginal value of a resource was applied to a typical HEA/REA scenario.

Zafonte and Hampton (2007) tested similar assumptions and found that HEA/REA performs well over a range of reasonable scenarios. For example, they tested the sensitivity of scaling results to various assumptions about changes in marginal value. Instead of assuming an annual two-percent change as suggested by Dunford *et al.* (2004), they examined likely changes in marginal value over time based on a utility-theoretic marginal value function and empirical data on changes in resource abundance. They also examined the issue of market scope, whereby those living close to a resource might place a greater value on marginal resource changes in light of a limited local supply. In both instances they found that ignoring differences in marginal value did not produce a significant effect except in extreme cases. In the majority of situations examined, the extension of the basic HEA/REA model to account for more precise economic assumptions did not change scaling results by more than 10 percent.

2.7 Economic concepts in ecological restoration

Understanding the importance of economic compensation and value in restoration scaling permits a critical evaluation of alternative scaling models. It also clarifies numerous conceptual issues that arise frequently in practice when determining appropriate compensation. Some of these issues are addressed next.

1. *The restoration concept of "nexus" can be defined rigorously in the context of public compensation.*

"Nexus" refers to the requirement that selected restoration projects have a close connection to the resource injury (e.g., NOAA, 1997). Relevant considerations include whether the selected projects constitute "in-kind" restoration, which directly enhances the same type of habitat or species that was injured. The proximity of restoration projects to the injured site is also an important consideration in evaluating nexus, with a distinction between preferred "on-site" projects and less preferred "off-site" projects. Judgments about what constitutes sufficient nexus are usually made intuitively based on the impressions of public officials and investigators.

In an economic context, the nexus requirement has a precise interpretation associated with providing compensation to the appropriate people. Specifically, nexus ensures that those who suffer losses from a resource injury are the same people who benefit from restoration. For example, the development of a sports facility as a restoration project may create significant value to the public, but there is no guarantee that the group of people who benefit from the facility is the same group that suffered losses resulting from a resource injury. People who place a high value on the loss of natural resource services may not participate in sports, but they are likely to benefit from natural resource enhancements. Resource enhancements would have greater nexus to the natural resource injury and would be more likely to compensate the appropriate people in a restoration scaling context.

Recognizing the role of public compensation can refine the way nexus is evaluated. If trout populations in an Oregon stream are reduced by contamination, ensuring the appropriate nexus may appear to require restoration in the same stream, or in another stream within the same watershed. However, if the people who use or otherwise place a value on the degraded stream live throughout a large rural area where numerous streams in many different watersheds are of equal importance, then the relevant population may be sufficiently compensated by restoration anywhere in the region. Geographic constraints may be relaxed to allow for selection of the most beneficial or least costly project in the area. Conversely, consider a contaminated river that is subject to a public advisory limiting the consumption of fish. If many people still like to fish on the river in spite of the advisory, then construction of a boat ramp allowing greater access to the river may be desirable to many people and may create considerable public value. Both the loss from advisories and the benefits from greater access are directly associated with recreational fishing, which may appear to suggest a strong nexus between injury and restoration. However, those most affected by the consumption advisory are unlikely to be those still fishing at the contaminated river. The people who incurred losses will therefore not benefit from the restoration action, and the nexus requirement is not satisfied.

2. *Objective measures of ecological service do not exist in the context of interim loss and judgment inevitably plays a role in determining appropriate compensatory restoration.*

Because value is subjective, there is no definitive method for evaluating the relative importance of various habitat components which may be affected in different ways following an incident. There is also no objective tradeoff between natural and created habitat, or between different types of habitat that may be enhanced to offset resource injury. Certain restoration scaling methods may appear to be more objective than others, but this is usually attributable to the use of implicit value assumptions in place of explicit ones.

Consider a swamp forest along a contaminated river in which bird populations have declined by 50 percent due to toxic effects. One approach to evaluating habitat loss would use bird populations as the metric, resulting in a 50-percent service loss over the affected area. This has the advantage of simplicity, relying solely on the parameter that has been explicitly measured. Another line of argument might suggest that birds are a small component of the overall habitat, which includes other forms of wildlife as well as plants and trees. Vegetation alone constitutes the majority of biomass in the swamp forest habitat, and the marsh may also provide filtration and flood control benefits. The importance of these various factors depends on people's preference for birds relative to other wildlife and water quality, and their preference for natural amenities like wildlife and water quality relative to human-use services such as flood control. Incorporating these factors into the analysis, a 50-percent decline affecting only birds might translate into, say, a 5-percent loss in total habitat services.

Alternatively, one might consider the tradeoffs implicit in the quantification of service loss. Suppose an area of swamp forest were to be restored and a location must be selected either in a contaminated area of the river or an uncontaminated area. In the contaminated area, river water would carry toxins into the marsh and the restored habitat would support a bird population that is 50 percent lower than normal due to toxic effects. Some people might have a strong preference for restoring the uncontaminated area and would be willing to accept a smaller area of restoration in return. This preference might be based on the view that birds are among the most attractive features of the habitat and should be give considerable weight when determining service loss. It might also be based on individual-specific factors such as a strong aversion to disturbance of the natural environment caused by the preventable release of manmade toxins. If people are willing to accept as compensation a clean marsh one-third the size of a newly created contaminated marsh, this would imply that a given area of contaminated marsh provides one-third the value of clean marsh. This equates to a service loss of 66 percent due to contamination.

In the initial example, no valid justification was presented for the choice of a 50-percent service loss. One could just as easily select a zero-percent service loss based on the lack of any measurable change in some alternative habitat parameter other than bird abundance. As soon as valid considerations are introduced into the exercise, such as the relative importance of various habitat components in the second example or the desire to avoid creating contaminated wetlands in the third example, the evaluation of service loss involves explicit judgments that are inevitably subjective. Recognizing the subjective nature of any valid approach to quantifying service loss is an important step in evaluating restoration scaling methods, because there is a tendency in practice to view with suspicion any method that makes subjective judgments an explicit feature of the analysis.

3. *Many issues pertaining to the measurement of resource services cannot be addressed based solely on the judgment of scientists because they are a matter of individual preference.*

Contamination could conceivably lead to deformities in fish larvae without causing detectable effects in adult fish. Contamination may cause liver tumors in adult fish without affecting the way fish behave, reproduce, or interact with the ecosystem. Fish mortality may be associated with contamination without causing a decline in fish populations because limiting factors such as food supply predominate. Biological expertise is required to identify and characterize these ecological conditions. However, there is no consensus among biologists as to whether each of these conditions constitutes an adverse ecological change for which compensation is required. Biologists disagree about what constitutes a loss of ecological service and how much compensation is required because these questions are a matter of preference. The opinions of biologists in these matters should not carry more weight in the NRD process than the opinions of other members of the general public. Valid resolution of these issues must be based on a reasonable representation of public preferences through the use of surveys or other means.

Similar questions arise in evaluating appropriate restoration actions. The restoration of native brook trout in the Great Smokey Mountains of Tennessee and North Carolina may be preferable to the restoration of non-native rainbow trout for many reasons. These include a precautionary concern regarding potentially unknown effects from the presence of non-native species, or a preference for native species over non-native species out of a fundamental appreciation for natural conditions that prevailed prior to human intervention. On the other hand, many anglers prefer to fish for rainbow trout and would be opposed to the choice of a restoration project for brook trout. The relative prevalence of those who fish and those who favor nature unaltered by human interference is likely to differ between the general public and any group that does not represent a random sample of the general public. If restoration decisions are left to solely to biologists and other experts directly involved in the NRD process, the selected restoration strategy is likely to reflect personal biases, which may result in too little, or too much, compensation of public losses. The fact that considerable public money is spent controlling some non-native species, such as sea lampreys in the Great Lakes, and no money is spent discouraging the spread of others, such as the common honey bee in North America (native to Asia and the Middle East), suggests that restoration strategies are appropriately determined by what people like or dislike about alternative natural environments.

The conclusion to be drawn is not that restoration scaling must always be based on surveys of the general public. In many cases the complexity of the ecological issues involved in evaluating injury and compensation may be significant enough that the use of public surveys would entail unacceptable levels of uncertainty in scaling outcomes. When scaling exercises such as HEA or REA are undertaken based on the judgment of scientists alone, economic principles demand that those involved in the scaling process avoid the unreasonable intrusion of personal bias and remain cognizant of the role of public value in compensatory scaling decisions.

4. *Restoration scaling methods based on ecological metrics measure the same service losses as methods based on public surveys.*

Since compensatory restoration scaling is based on value to the public, metric-based models such as HEA or REA and economic models based on stated-preference surveys should in principle obtain the same results. Of course, random uncertainty regarding the inputs for both methods can lead to a divergence of results in any given instance. More fundamentally, it is useful to consider two important sources for any discrepancy between metric-based methods and public surveys. The first is a difference in preference between practitioners developing HEA/REA models and members of the general public. If preference is the determining factor in the selection of a particular scaling input, then the selection should be made based on information from surveys of the general public. Alternatively, discrepancies between metric-based methods and surveys of the general public may arise due to a lack of understanding of on the part of the public regarding the relevant scientific principles. In this case, the most valid source of scaling inputs is likely to be the judgment of practitioners or a panel of scientific experts.

In some cases, metric-based models can be informed by a limited investigation of public preference. For example, focus groups were used to investigate preferences for the enhancement of naturally spawning salmon as compared to the use of salmon stocking programs as compensation for fish mortality at the Blackbird Mine site in central Idaho (Chapman et al. 1998). An assessment of public preferences helped determine the appropriate weights assigned to the restoration of alternative resource services in a Puget Sound estuary (Commencement Bay Trustee Council, 2002). The improvement of ecological metrics using information from public surveys represents an intriguing avenue for further interdisciplinary research.

2.8 Stated-preference methods for ecological scaling

Compensation for interim losses must make members of the public as well off after an incident as they would have been had the incident not occurred. In the preceding sections we described some of the limitations that arise when using ecological metrics to determine public compensation. Fundamental among these limitations is that appropriate compensation depends on individual preference, and the preferences of individuals affected by an incident are not known to the scientists who evaluate the appropriateness of ecological scaling metrics.

Stated preference methods use surveys of the general population to address the importance of public preferences in restoration scaling. The surveys usually inform respondents about an environmental injury and ask respondents how much they would be willing to pay to prevent such injury in the future, or how much resource restoration they would accept as compensation for the injury. The advantage of stated preference methods is their strong connection to economic principles and valid statistical methods. If compensation is adequate for the public as a whole, it should be possible to demonstrate that compensation is adequate for a random sample of the public. Surveys attempt to accomplish this by drawing on techniques used in commercial marketing, pharmaceutical research and development, political analysis, policy research, and cost-benefit analysis. In the public policy arena alone over 5,000 studies in more than 100 countries have used surveys to value public goods (Carson, forthcoming).

The drawback of survey-based methods is the potential difficulty respondents may have in evaluating public resource issues. While some researchers suggest that people cannot have a value for goods unknown to them (Johnson *et al.,* 2001), it is precisely a lack of specific knowledge about valued goods that makes stated preference difficult. Having entrusted public officials with the task of stewardship over commonly held resources, people have every incentive to let their attention lapse regarding details of how public budgets are spent and how environmental quality is maintained. This potential lack of familiarity regarding the range of goods available and alternative uses of public money leads to some of the fundamental critiques of stated-preference methods. The methods are nonetheless widely applied, as the questions they address become increasingly important and the techniques continue to be improved.

There are numerous published sources providing details on stated preference methods. Mitchell and Carson (1989) was the first comprehensive treatment and continues to be a primary resource. Bateman and Willis (1999) presented an update of important developments summarized by many leading authors in the field. Practitioners often rely on Haab and McConnell (2003) and Train (2003) for a detailed understanding of the relevant econometric techniques. Louviere *et al.* (2000) and Kanninen (2007) focus on concepts and techniques for choice experiments, a modern approach to stated preference. Champ *et al.* (2003) is a good overview of stated preference and other current methods in resource valuation.

The following section does not attempt to review methodological details that are presented elsewhere, but instead focuses on two objectives. The first objective is to provide a basic familiarity with topics relevant to government officials, scientists and attorneys working alongside experts in the field. The second objective is to highlight studies and findings specific to the field of NRD. We begin with a discussion of monetary valuation, which was the original focus of stated-preference methods and is still the objective in most survey-based research on public goods outside the field of NRD. When applied to NRD, monetary valuation does not lead to a precise scaling of restoration projects. Instead, funds are collected which are equivalent to the estimated loss in public value, and resource trustees select restoration projects based on the amount of the available funds. An alternative to this approach is based on recent developments involving choice experiments, which evaluate resource options based on tradeoffs between specific resource attributes. Choice experiments are currently the favored method for NRD assessment, because the scaling of resource restoration is explicitly incorporated into the choice-experiment format.

2.8.1 Monetary losses

The natural resource damage assessment for the Exxon Valdez oil spill resulted in a $900 million settlement and sparked intense scrutiny of methods for valuing resource damages. Considerable controversy arose over the assessment of "passive-use" value, which refers to value the public holds for the existence of natural resources apart from their use in human activities like recreation or navigation. Though all methods for ecological scaling are fundamentally based on passive-use value, methods such as HEA and REA were developed to sidestep some of the controversy by avoiding explicit monetary valuation of the environment (Unsworth and Bishop, 1994; NOAA, 1996). The increasing use of a choice-experiment approach to stated preference, covered in the next section, also deemphasizes the importance of monetary tradeoffs. The more traditional economic methods that examine resource preferences based solely on tradeoffs between money and the environment are usually identified by the term "contingent valuation".

Despite the shift away from monetary losses and contingent valuation in damage assessment, we address this topic for two reasons. First, the approach is still used in NRD and it may be the most appropriate method in some circumstances. For example, it may be desirable to proceed with an assessment and settlement of a claim prior to a complete investigation of available restoration options. Second, many of the issues that arise in any type of stated-preference exercise have been most thoroughly investigated in the context of contingent valuation. Basic issues presented in this section are relevant to other survey-based methods for the assessment of ecological losses, even if other methods may address certain issues more effectively.

The central controversy over contingent valuation methods relates to their validity and accuracy. Side-by-side articles by Diamond and Hausman (1994) and Hanemann (1994) illustrate the alternative perspectives. Perhaps the greatest concern involves the issue of hypothetical bias. Because respondents are typically not required to make the payments presented to them in survey questions, there may be a discrepancy between people's true willingness to pay and willingness to pay as expressed in survey responses. Many studies have attempted to evaluate hypothetical bias indirectly by examining goods for which real and hypothetical transactions can be compared. The evidence is necessarily indirect because of the absence of any actual markets for environmental goods. List and Shogren (1998) compared surveys measuring willingness to pay for particular baseball cards to amounts people actually bid for the cards in a real auction. Cummings and Taylor (1999) compared hypothetical contributions to a charitable organization expressed in survey responses to actual charitable contributions. The results of 28 such studies were analyzed by Murphy *et al.* (2005), and the authors concluded that the median ratio of hypothetical to actual values was 1.35. The average ratio was 2.6 due to a skewed distribution that included a small number of exceptionally high values.

The degree of bias measured in these studies does not fully capture the effect of more recent techniques developed to mitigate hypothetical bias. Cummings and Taylor (1999) suggested that survey respondents should be directly confronted with the issue of hypothetical bias. In what is referred to as the "cheap talk" method of addressing bias, surveys include statements urging respondents to refrain from expressing a willingness to pay for a good if they do not mean it. Others have found that cheap talk indeed reduces overall estimates of willingness to pay, but not in a way that suggests more valid results. For example, it may induce respondents to reject purchases in an arbitrary and inconsistent manner (List *et al*., 2006). Several authors have found that hypothetical bias can be attributed to uncertainty on the part of certain respondents regarding their expressed preference (Champ and Bishop, 2001; Poe *et al.,* 2002; Champ *et al.,* 1997). By asking respondents to rate their level of certainty on a scale of 1 to 10, those who are less certain of their responses (with a rating below seven, for example) can be eliminated from consideration when computing willingness to pay. After the adjustment, hypothetical payments appear to reproduce actual payments in side-by-side comparisons.

Another important consideration in addressing hypothetical bias involves respondent perceptions about whether the results of a survey have real consequences (Cummings and Taylor, 1998). In other words, hypothetical bias is reduced if surveys appear less hypothetical. The consequential nature of survey questions is really a prerequisite for applying standard economic theory (Carson and Groves, 2007). The importance of "consequentiality" in communicating the relevance of survey questions can be illustrated by comparing a terse elicitation format like "What is the most you would be willing to pay for X?" to a thorough scenario for payment and provision of a proposed environmental good (e.g., Carson *et al*., 2003, discussed below). Some studies have specifically identified an inverse relationship between hypothetical bias and the perceived probability by respondents that their survey choices could lead to implementation of an environmental program (Cummings and Taylor, 1998; Mitani and Flores, 2008).

A scope test is meant to address another fundamental aspect of the validity of survey results, and most contingent valuation studies include a scope test. The premise is that willingness to pay for environmental improvements should be sensitive to the extent of the improvements; that is, willingness to pay should increase with the scope of the proposed action. Boyle *et al.* (1994) famously found that willingness to pay to prevent the death of birds in the central flyway of the United States did not change when the specified number of deaths prevented varied from 2,000 to 20,000 to 200,000. To test for sensitivity to scope, surveys typically present at least two different levels of the environmental good being offered to two distinct groups of respondents. This allows for a statistically valid analysis of the response of willingness to pay to changes in scope.

Choices presented in a survey must make sense to respondents and must be consistent with any previous knowledge respondents may have. If this is not the case, surveys may not obtain valid information on respondent preferences due to scenario rejection. For example, people might be reluctant to agree to higher taxes in order to fund water-quality improvements at a lake if they know that companies responsible for contaminating the lake are supposed to be held liable for cleanup and restoration. Stated willingness to pay may understate true willingness to pay due to rejection of any scenario in which public funds are used for cleanup. In this case scenario rejection may be mitigated by suggesting that some costs will be borne by industry, but that additional funds will be required from public sources (Bishop *et al.,* 2000). Scenario rejection in NRD surveys can also arise due to concerns people may have about cleanup operations, such as dredging a river or lake. If the scenario presented to respondents involves willingness to pay for removal of environmental contaminants, their expressed values will account for any negative perceptions about the cleanup process. This may lead to an underestimate of natural resource damages, which should be based solely on willingness to pay for the absence of contamination regardless of cleanup costs.

The term "protest" refers to certain types of responses that do not represent an honest expression of the respondent's willingness to pay. Protest can be related to scenario rejection, as in the above example where respondents wanted the RP (Responsible Party) to pay and were not willing to reveal their own willingness to pay. Protest responses in essence reflect a refusal to participate in the valuation exercise. In most cases protest is associated with respondents who answer all questions as if their value for the proposed resource improvement were zero. Protest zeros are typically distinguished from true zeros using follow-up questions, where respondents give a reason for their choice. Following the above example, a zero willingness-to-pay response may be followed by a comment stating that "the responsible party should pay to clean up the lake". By contrast, the comment "I pay enough in taxes" might indicate protest against a proposed increase in taxes to pay for environmental improvements. However, the comment is also consistent with a zero willingness to pay for improvements and would not be recorded as protest. Protest may also take the form of responses suggesting an infinite willingness to pay, rather than zero willingness to pay. For example, the comment "one cannot put a price on the environment" suggests that a respondent may consistently choose the highest possible level of environmental improvement without proper consideration of limits in personal income, or other constraints.

The selection of an appropriate payment vehicle is an important part of survey design. The payment vehicle refers to the mechanism for collecting money in the context of the survey scenario. For ecological issues, an increase in tax payments by individuals or households is a common payment vehicle (Carson *et al*., 2003; Bishop *et al.,* 2000). Ecological issues generally affect a population that is broader than just those who visit a site for recreation, so user fees and travel costs cannot be applied as payment vehicles in ecological studies. When describing an increase in taxes, the researcher must choose whether it is more appropriate to use a one-time payment or an annual payment over several years. Stevens *et al.* (1997) suggested that the choice of time frame for payments can lead to a significant divergence in willingness-to-pay estimates compared to expectations based on standard discounting procedures. The use of a one-time payment may be preferred because it tends to result in more conservative willingness-to-pay estimates (Carson *et al.,* 2003). Other examples of payment vehicles include an increase in utility bills for groundwater improvements (Powell *et al*., 1994) and an increase in prices for fuel and consumer products in the context of climate change (Layton and Brown, 2000). Charitable contributions have also been used as a payment vehicle (Champ *et al*., 1997) but the resulting value estimates are considered to be a lower bound due to the potential for "free-riding".

Free-riding is one of the issues considered under the topic of "incentive compatibility" in contingent valuation. Charitable contributions are not incentive compatible because people have a strong motivation to free ride, that is, to enjoy the benefit of any programs funded by other people's donations while themselves making no contribution. This is in contrast to a voter referendum, in which everybody is required to pay if the referendum succeeds and there is no way to save money by voting no. Referenda are viewed as incentive compatible because people have an incentive to vote yes if and only if they support the proposed action at the required cost. This is one of the primary reasons the "dichotomous-choice referendum" format is the most common approach to stated preference surveys. The referendum format calls for a yes or no response to a specified environmental change presented to respondents at a particular price. A recent overview of incentive compatibility is presented in Carson and Groves (2007). Any contingent valuation study in the NRD arena is likely to follow this format, following recommendations of the NOAA Blue Ribbon Panel that examined the use of contingent valuation in NRD (Arrow *et al*., 1993).

To evaluate willingness to pay, different respondents to a survey are presented with alternative costs for the specified environmental change. This allows the analyst to develop a distribution of willingness to pay which can be applied to members of a heterogeneous population. The costs presented to respondents are often called "bids", and "optimal bid design" is the investigation of appropriate levels of bids to be used in a study. The bids must vary sufficiently to identify people with both low and high values, while not squandering questions that propose payments too high or too low to be relevant for most respondents. Important articles addressing aspects of bid design include Cooper (1993), Alberini (1995a,b), and Kanninen (1995).

We briefly describe three studies that illustrate the use of contingent valuation in NRD assessments. The Exxon Valdez oil spill study is summarized in Carson *et al.* (2003), and the Montrose study assessing contamination in the Los Angeles harbor is presented in Carson *et al.* (1994). An extensive case study of contingent valuation for oil spills was funded as part of the settlement of the 1988 Martinez oil spill in northern California, and the details of the case study appear in Carson *et al.* (2004).

Losses from the Exxon Valdez spill were evaluated based on the public's willingness to pay for a program that would prevent a similar spill in the future (Carson *et al*., 2003). The program consisted of escort ships that would guide vessels through navigation channels in Prince William Sound, the site of the Exxon Valdez spill. The challenge was to describe the circumstances of the spill and impacts to the environment in an accurate and complete way, while relying on materials that would be comprehensible to respondents with a wide variety of education levels and life experiences. As in most stated-preference studies, developing the appropriate materials required a succession of focus groups conducted with a sample of potential respondents to determine how they perceived and evaluated the issues under investigation. Focus groups were followed by pretests, in which potential respondents completed an initial draft of the survey and were debriefed by researchers in one-on-one interviews. The final survey was developed based on findings from the focus groups and pretests. Once complete, the survey itself was administered using in-person interviews with a sample of respondents drawn from throughout the U.S. In-person interviews are considerably more expensive than mail or telephone surveys, but are thought to ensure the most valid results.

The Exxon Valdez survey employed nineteen visual aids, including maps and pictures, to convey the relevant information. The types of birds and other wildlife affected by the spill were presented in pictures, but in order to be conservative, no pictures of oiled wildlife were shown. Also, assurance was given that none of the twelve bird species most affected by the spill was threatened with extinction because of the spill, and respondents were informed that large bird kills can occur naturally. It was important that the use of escort ships to prevent or contain a future spill be perceived as feasible, effective, and requiring the amount of money presented as the cost of the program. The survey materials emphasized that the program would not provide spill protection outside Prince William Sound. This last point was meant to address any potential "embedding", whereby respondents may falsely perceive that a proposed program addresses a broad class of problems related to the specific issue under consideration.

To investigate potential scenario rejection, respondents were asked whether they felt that the proposed program would only partly contain damage in a future incident, or would not do anything to prevent damage. Responses to these questions were added as variables to the model of willingness to pay. This gave researchers the option to adjust for scenario rejection by switching off these variables during the analysis phase, thus estimating willingness to pay as if all respondents accepted the effectiveness of the program. Another variable addressed protest votes in a similar way. Specifically, the model estimated the decline in willingness to pay associated with those who felt that Exxon should pay rather than the public. This decline in value could be added back in when estimating damages in the analysis phase of the study.

The claim for damages in the Exxon case was developed without these adjustments. The median value of losses was $30 for each English-speaking household in the U.S., resulting in a total estimated loss of $2.8 billion in 1990 dollars. While the Exxon Valdez incident also caused modest losses to recreation, without the assessment of passive-use losses the total estimate of damages from one of the most significant environmental incidents in recent times would have been only a few million dollars. The authors of the Carson *et al.* (2003) study argued that recognition of passive use losses in the Oil Pollution Act of 1990 may have been responsible for the decline large oil spills in the U.S. relative to elsewhere in the world. International law governing oil spills has generally not recognized liability for passive use losses.

The other major precedent for the use of contingent valuation in NRD involved the Montrose assessment (Carson *et al.,* 1994). The Montrose contingent valuation study was based on a referendum format that proposed a one-time tax to all California households in exchange for a program to accelerate the natural decay of DDT and PCB contamination in Los Angeles harbor. The survey described reproductive difficulties of certain species of birds and fish due to contamination and indicated that without the program these effects would persist for 50 years, but that with the program recovery would be achieved in five years. Unfortunately, the conclusions of scientific studies that demonstrated the effects on birds and fish were modified during the course of the investigation and legal proceedings. This made the economic valuation study difficult to interpret, given the specific description of ecological injuries to be remedied and the take-it-or-leave-it premise of the referendum format. The Montrose case thus illustrates one advantage of choice experiments, in which the focus of valuation is on specific attributes of an ecosystem that could potentially be adjusted to reflect the revised scientific conclusions. Even with the flexibility of attribute-based methods there would be limitations in the way scientific information is presented, and the evaluation of alternative injury scenarios may require the use of a split sample with different respondents evaluating different potential circumstances.

Carson *et al.* (2004) used funding obtained from the Martinez settlement to investigate the value of damages that could result from potential oil spills on the California coast. Their approach is similar to the Exxon Valdez study in that a program is proposed to prevent spills using vessel escorts. A key contribution of the published study is the response it offers to a critique of the contingent valuation method presented in Dunford *et al.* (1996). Together the two documents represent the type of exchange that characterizes the debate in NRD assessments over the use stated-preference methods for ecological losses. The issues include alternative perspectives on hypothetical bias and testing for scope. The Carson *et al.* (2004) study also discusses aspects of respondent comprehension and motivation, citing written comments by survey respondents and illustrating the type of scrutiny that takes place over the course of a damage assessment at a level of detail rarely available in the published literature.

2.8.2 Resource tradeoffs

Controversy over contingent valuation combined with the convenience of HEA has led some practitioners to abandon survey-based restoration scaling, but recently interest in survey methods has been revived by the development of choice experiments that evaluate resource tradeoffs. Choice experiments alleviate many of the methodological difficulties with contingent valuation, and can avoid explicit monetary valuation of losses by directly determining the amount of resource restoration required to offset losses. Some of the first studies that applied choice experiments to environmental resources include Adamowicz *et al.* (1998) and Hanley *et al.* (1998).

Instead of voting "yes" or "no" to a specified policy at a particular price, respondents to a choice experiment select their preferred option from among bundles of alternative attribute levels. For example, option A might offer a large amount of protected habitat and a modest program to control agricultural runoff, while option B offers less habitat protection but greater runoff control. If the specified attributes include the level of contamination at an affected site, choice experiments can determine the amount of restoration respondents would be willing to accept as compensation for injuries due to contamination. "Choice experiment" appears to be the most common term for this approach in the economics literature, but the terms "conjoint analysis", "stated choice", and "attribute-based methods" are also used to refer to this type of approach. The term "conjoint analysis" is borrowed from the psychology and marketing literature and refers to the fact that choices involve numerous attributes that are "considered jointly". An illustration of the choice experiment format is provided in the discussion of the Green Bay, Wisconsin damage assessment below.

When choice experiments are used to directly determine appropriate compensatory restoration, problems of hypothetical bias may be largely avoided. Fundamentally, hypothetical bias reflects the tendency of hypothetical monetary payments to exceed what people are willing to pay in actual cash transactions. Once monetary payments are eliminated, it would seem unlikely that any systematic influences would tend to bias resource tradeoffs in such a way that damage estimates are consistently too high or too low relative to people's true preferences. Even when monetary payments are included among the choice attributes, any tendency to overstate willingness to pay for one environmental improvement (such as contaminant removal) is likely to cancel out when compared to the value of another improvement (such as resource restoration projects). Furthermore, there is some evidence that attribute values estimated in choice experiments may not be influenced by hypothetical bias. List *et al.* (2006) found that hypothetical payments exceed actual payments when using choice experiments to evaluate a policy as a whole, but not when evaluating the marginal values of particular policy attributes. This may be a consequence of the fact that "yea-saying", or expressing support for a proposal even when the specified dollar amount is too high, is less likely to be a temptation with regard to marginal attribute values (Adamowicz *et al.*, 1999a). This follows partly because respondents are not limited to a single up or down vote as in contingent valuation, but are instead presented with a range of payment and attribute levels. Respondents can express a positive willingness to pay at some attribute and payment levels, while refusing to pay in other instances when the required payment is too high.

The greatest advantage of choice experiments in NRD may be the ability to develop a restoration-based damage claim. The process of determining appropriate restoration and then developing a damage claim based on the cost of the required restoration projects is preferred to using lost value as the basis for a claim (NOAA, 1996). First, while funds equivalent to lost value would be sufficient to compensate the public using monetary payments, there is no guarantee that restoration undertaken with the collected funds provides sufficient compensation. Apart from any uncertainty in methodological results, restoration-based claims ensure that the public is appropriately compensated. Second, estimates of lost monetary value depend significantly on the size of the population studied. If the Exxon Valdez study were confined to Alaska rather than the whole U.S., or extended to Canada and other parts of the world, calculated damages would have changed significantly. This potentially makes damage estimates highly sensitive to researcher discretion. By contrast, an estimate of restoration and resource tradeoffs may be refined when additional populations are surveyed, but there is no direct relationship between the damage amount and the number of people included in the study population. Finally, complications associated with discounting and the timing of costs and benefits can be avoided using a restoration-based approach. While the selection of an appropriate payment vehicle involves consideration of the timeframe over which payments and policy actions occur (Boyle, 2003), most environmental improvements occur over similar medium-term or long-term horizons that can be reasonably compared without significant discounting uncertainties.

Another advantage of choice experiments involves the issue of scope. While testing for sensitivity of willingness to pay to the scope of a proposed improvement is explicitly considered in contingent valuation, the issue is addressed implicitly in choice experiments. In choice experiments respondents are presented with alternative levels of environmental attributes as part of the choice design. Lehtonen *et al.* (2003) found that willingness to pay for land preservation in Finland was not sensitive to scope when contingent valuation methods were used, but that scope mattered in a choice-experiment study of the same issue. This and other studies suggest that confronting respondents with alternative levels of attributes may generate more thoughtful and accurate responses. However, the divergence of results can also be interpreted as a lack of consistency between methods, that is, a failure of what is known as "convergent validity". There is also evidence that sensitivity to scope in choice experiments, while apparent for a given respondent choosing among alternative attribute levels, is less apparent across respondents presented with different ranges of attribute levels (Bateman *et al.*, 2007; Luisetti *et al.*, 2008).

The complexity of choice experiments relative to contingent valuation is another potential complication of the resource-tradeoff approach. Research indicates that choice experiments place a cognitive burden on respondents that can lead to biased results. DeShazo and Fermo (2002) found that an increase in the number of attributes presented to respondents and an increase in the number of attributes that are allowed to differ across alternatives in a given choice question both lead to greater randomness in respondent choices. In response to task complexity, respondents may employ simplifying strategies. For example, Hensher *et al.* (2005) found that many respondents ignored certain attributes when coping with complex choice tasks. Using follow-up questions to identify which attributes were ignored by particular respondents, they were able to incorporate these simplified choice strategies into the model by setting the appropriate attribute coefficients to zero. They found that the standard approach to choice experiment analysis, based on the assumption that all attributes influence choice behavior, may lead to biased results.

The potential complexity of choice experiments also forces researchers to impose simplifications on the design of choice questions prior to implementation of a survey. Experimental design refers to the selection of particular combinations of attribute levels to be presented to respondents. For example, if each of the five attributes in Figure 2 is evaluated at three different levels, then there are $3^5 = 243$ different versions of each alternative. The number of choice questions representing a unique comparison between two alternatives is much greater. The decision of which questions to include in a survey has implications for the estimation and interpretation of model results. One simplification often invoked in survey design is the assumption that the marginal value of a change in each attribute is independent of the level of other attributes. This results in a "main effects" design for selecting survey questions, and it dramatically reduces the number of choice questions required for estimating an econometric model. It is also common to expand a main effects design to allow for two-way interactions between particular attributes. Methods for experimental design are described in Holmes and Adamowicz (2004) and Johnson *et al.* (2007).

A theoretical drawback of choice experiments is that they do not meet the requirements of incentive compatibility, as described in Carson and Groves (2007). The problem arises because of the multiple alternatives incorporated in the choice-experiment format. In a simple binary referendum format, respondents choose between the status quo and the specific policy or program presented to them at a given cost. It is theoretically possible to convince respondents that a yes vote will make implementation of the program more likely, and a no vote will make implementation of the program less likely. Under such circumstances respondents are motivated to vote for the program if and only if they favor the program at the stated cost. Thus the incentive structure encourages respondents to reveal their true willingness to pay.

In the choice-experiment format, respondents are presented with a tradeoff between two or more options in each of several choice questions. Given the range of outcomes under consideration, respondents can only speculate about how their choices will be used to determine policy and what outcome will result. Some respondents may assume that the alternative receiving the most votes overall will be chosen by policy makers. This could lead a respondent to vote against her most preferred option in favor of, say, her second-most preferred option, if she believes her second-most preferred option is more likely to get enough total votes to be selected. Even if the respondent is an expert in economic modeling and correctly understands the use of the surveys, she may be motivated to misstate her preferences in one direction or the other in order to offset the potential opposing views of other respondents. The degree to which such theoretical considerations affect choice-experiment studies in practice is a matter of debate.

The most significant NRD study using choice experiments was performed for the Green Bay damage assessment, including waters in Wisconsin and Michigan. The study was conducted for the U.S. Fish and Wildlife Service (Bishop *et al*., 2000) has since been summarized by Breffle and Rowe (2002) and Lazo *et al.* (2005). An example of the choice questions used in the survey appears in Figure 2.1.

	Alternative A ▼	Alternative B ▼
Wetlands Acres	58,000 acres (current)	58,000 acres (current)
PCBs Years until safe for nearly all fish and wildlife	100+ years until safe (current)	40 years until safe (60% faster)
Outdoor Recreation Facilities at existing parks	10% more	0% more
Acres in new parks	0 acres (current)	0 acres (current)
Runoff Average water clarity in the southern Bay	34 inches (70% deeper)	20 inches (current)
Excess algae days in lower Bay .	40 days or less (50% fewer)	80 days or less (current)
Added cost to your household Each year for 10 years	$50 more	$50 more
Check (✔) the box for the alternative you prefer →	❑	❑

Figure 2.1. Choice-experiment for the Green Bay damage assessment.

The injuries in the Green Bay case were caused by PCBs, which were determined to be responsible for reduced reproduction rates in several bird species, liver abnormalities in walleye, and potential effects on other animals, such as mink. Fish consumption advisories were also issued for PCBs. These natural resource injuries were described in the introductory materials for the survey. As shown in Figure 2.1, respondents were offered the choice between a policy that would leave current conditions unchanged, described as “100+ years” until the resource is safe for wildlife, and another policy that would reduce to 40 years the time until the resource is safe for wildlife. The second policy involved the use of dredging to remove significant contamination from Green Bay and Fox River, a tributary of the Bay. The change in PCB injuries was also expressed in percentage terms: specifically the improvement from PCB removal was described as a 60-percent reduction in the time until the resource is safe for wildlife. Using percentage changes to express a policy's effect on resource attributes and assist with interpretation by respondents is common in the literature on choice experiments (Horne and Petäjistö 2003; Colombo *et al.* 2005).

The Green Bay survey examined tradeoffs between PCB removal and three potential restoration options, including restoration of wetlands, control of agricultural and urban runoff, and improvements to recreational facilities. The benefits of these programs, such as improved fish spawning due to wetland restoration and improved water clarity from runoff control, were described in the survey materials. The restoration benefits were quantified for the choice tradeoffs in terms of total wetland acres in the region, acres of new parks for recreation, and average water clarity in Green Bay, expressed in inches. Consistent with the description of injury, the proposed restoration improvements were also expressed in percentage terms. The payment vehicle was described as an additional cost to each household, to be collected through federal, state, and local taxes. Presenting payments as a cost to households rather than to individuals may result in lower estimates of total willingness to pay (Lindhjem and Navrud 2008).

3.0 Human Use Losses

A significant part of the value of natural resources derives from their use in human activities. Restoration scaling methods address resource losses associated with recreation, navigation, subsistence harvest, the cultural activities of aboriginal people such as American Indian tribes, or other purposes. The most common type of human-use assessment involves the analysis of recreational activities affected by oil spills and toxic contamination. Revealed-preference methods use observed changes in the demand for recreation trips to estimate changes in value, while stated-preference methods rely on hypothetical choices presented to respondents in a survey. There is also increasing interest in addressing the cultural importance of natural resources to native tribes when assessing losses on tribal lands. While revealed-preference methods have been used to evaluate tribal losses, the focus of recent research is on stated-preference surveys, focus groups, and other interactive evaluation methods that are sensitive to cultural issues. Finally, this section describes assessment methods associated with impacts to commercial navigation and price increases for market goods, which may represent lost value to the public under certain circumstances.

3.1 Value to cost scaling

The most common scaling method for human-use assessments relies solely on monetary valuation of losses, whereby funds equivalent to monetary losses are recovered and applied to restoration projects. This "value to cost" approach to scaling is only an approximation because it does not ensure that the selected restoration projects provide sufficient compensation for losses. Alternatively, the benefits of restoration projects may be directly evaluated so that a suite of projects can be specifically selected to offset lost value. This latter approach is analogous to the equivalency models for ecological services described previously and is sometimes called "value to value" scaling.

The drawback of value-to-value scaling for human-use assessments is that resource enhancements intended to benefit human uses also provide numerous ecological services. For example, it would be possible to offset contaminant-related losses to anglers using restoration of spawning habitat for fish. The increase in fish populations would improve catch rates, which would provide value to anglers. However, the benefit to anglers through increased catch rates is likely to almost undetectable, while the value people place on the restored habitat could be significant. A similar problem arises when restoration projects specifically target human uses. A loss to anglers may be offset by the construction of a new boat ramp, but construction of the boat ramp would displace shoreline habitat and could increase congestion and human disturbance in nearby natural areas. These indirect losses to both anglers and the general population would need to be explicitly addressed in a value-to-value scaling exercise.

To avoid these complicating factors, most methods described below take advantage of the simplicity of value-to-cost scaling. Because value-to-cost scaling involves a departure from the standard model that equilibrates losses and gains, it is important to describe the underlying justification for this common scaling approach.

3.1.1 Justification for value-to-cost scaling

In value-to-cost scaling, restoration projects are selected such that the cost of restoration is equivalent to the value of resource losses. In order to make the required connection between value lost and value gained, the value-to-cost approach relies on the premise that the selected restoration projects would pass an appropriate cost-benefit test. For example, if a project to combat beach erosion passes a cost-benefit test, then the benefits to beachgoers provided by the project must be at least as great as the project's cost. If the project is selected in the context of value-to-cost scaling, the project's cost would be equivalent to the value lost from a resource injury. The implied relationship between value lost, cost expended, and value gained suggests that the value of the project is at least as great as the loss for which compensation is required. Value-to-cost scaling thus ensures that the selected restoration projects are sufficient to compensate for spill-related losses as long as the projects are worth doing in the first place.

Spill-related restoration is undertaken in coordination with local resource managers. These officials do not usually evaluate projects of modest scale using rigorous cost-benefit analysis. Instead they make judgments based on knowledge of local resource conditions and observed public demand for resource use. If managers in charge of public spending significantly overvalue the benefits of resource projects, then value-to-cost scaling could lead to insufficient compensation. This is because the benefits of selected restoration projects may be less than their cost and therefore less than the assessed value of losses. Alternatively, if public funding for resource-related projects is typically below optimal levels, then value-to-cost scaling could lead to excessive compensation. This is because projects with benefits exceeding their cost would remain unfunded and would be available for use in offsetting natural resource damages, producing benefits greater than assessed losses.

In general, one expects that resource managers evaluate public projects appropriately and that public spending priorities do not deviate too greatly from the optimal level. Even if many people argue that one or the other of the above conditions prevails and a given project is over- or undervalued, this would not necessarily present difficulties for value-to-cost scaling. Restoration scaling methods may fail to satisfy the preferences of any particular individual, while still providing a reasonable approximation to average preferences.

3.2 Measuring lost value

This section introduces methods used to evaluate recreational losses. For methods and concepts specific to NRD and restoration scaling, we provide detailed explanations not available elsewhere in the literature. For valuation techniques commonly used throughout resource economics, we provide an intuitive introduction for NRD practitioners. Additional detail on the relevant valuation techniques can be found in Herriges and Kling (1999), Champ *et al.* (2003), and elsewhere. The topics addressed below include the distinction between revealed and stated preference, the meaning of "willingness to pay" and "consumer surplus", the way a travel-cost model works, methods and issues associated with stated-preference methods, and benefit transfer approaches relevant to recreation valuation.

3.2.1 Stated preference vs. revealed preference

Economists analyze value using information on what people say and what people do. The first approach is called stated preference and the second is call revealed preference. Stated preference data are obtained from surveys asking people to respond to hypothetical choices. This type of survey was discussed in the context of valuing ecological services, but similar methods can also be used to value human-use services. For example, a survey could be sent to anglers asking whether they would support a program costing $10 in additional license fees and leading to improvements in recreational fishing, such as removal of contaminated sediments from a local river. Revealed preference methods also rely on surveys to collect information, but the survey questions involve activities and choices that have already taken place. For example, anglers might be asked to report the number of fishing trips they took in a given month and which fishing sites they visited. These choices reveal the value of resources to anglers because they involve a tradeoff between fishing opportunities and monetary expenses, such as the cost of driving to particular fishing sites.

The advantage of revealed preference methods is that collection and interpretation of data is relatively straightforward, since it simply involves a record of individuals' recent outdoor activity. The most significant data-related issue is "recall bias" arising from limitations in the ability of respondents to precisely recall and report their activities. By contrast, the collection of data for stated-preference studies involves an extensive process of survey development in which choice questions are carefully crafted to present realistic scenarios, avoid ambiguities, and simplify potentially complex choices. The advantage of stated preference methods is their ability to investigate issues that do not arise in real-world choices. For example, it is not possible to obtain revealed-preference data on the choice an angler would make between a clean New York Harbor and a contaminated New York Harbor because only current conditions (a contaminated New York Harbor) are represented in observed choices.

For NRD assessments of lost human use, the current practice relies primarily on two resource valuation models: travel-cost models and choice experiments. A travel-cost model is a revealed-preference method based on the demand for trips to sites that are used for recreational or cultural activities. The term "random utility model" is often used, reflecting the economic theory that multi-site travel-cost models are based on. Choice experiments are a stated-preference method based on the demand for particular resource attributes associated with outdoor activities. Respondents choose between two or more hypothetical resource conditions, each presented as a collection of site attributes. The terms "conjoint analysis", "stated choice", and "attribute-based methods" are sometimes used interchangeably to refer to this type of approach. The somewhat cryptic term "conjoint" comes from the psychology and marketing literature and refers to the fact that choices involve numerous attributes that are "considered jointly".

It is possible to combine revealed-preference and stated-preference methods, and this is often viewed as the most valid approach. The combined model derives most information about site choice and the overall value of outdoor activities from revealed-preference data. Information about key attributes of interest, such as the presence of contamination at a site, usually comes from both revealed-preference and stated-preference data.

3.2.2. Willingness to pay and consumer surplus

"Consumer surplus" is the most common measure of value in the assessment of human-use losses. Consumer surplus refers to the value of goods or services net of expenses incurred to obtain them. It is also described as peoples' maximum willingness to pay for goods or services less what they actually pay.

To understand consumer surplus, it is worth distinguishing between value and price. Value represents what something is worth and is determined by individual preference. Price represents the cost of acquiring something and is determined by factors external to the individual. Many people unfamiliar with economic concepts mistakenly equate value and price, probably because price is the most relevant monetary measure in many familiar situations. For example, the relevant loss associated with a stolen bicycle is the bicycle's price, because the cost of replacing the bicycle is the price of a new one. Unfortunately this situation obscures the fact that the bicycle may have a value considerably higher than its price. The true value of the bicycle is not important to the owner of a stolen bicycle, because whatever the value is, it will be replaced when a new bicycle is purchased.

In other situations the importance of value is more apparent. Consider an individual deciding whether to purchase a bicycle in first place. In this case the relevant considerations include not only the price of the bicycle, but also the value derived from owning the bicycle. If the value is greater than the price, the individual should make the purchase. In other words, the individual should be interested in the "surplus" value above and beyond price, or what is known as consumer surplus. The purchase is worthwhile if consumer surplus is greater than zero. Consumer surplus is an essential concept in economic analysis because it accounts for both the costs and benefits of economic choices.

Consumer surplus is the relevant concept for assessing natural resource damages because it represents the net loss to the public when resources are impacted by pollution. For example, when people cannot take trips to the beach due to a beach closure, they lose the pleasure of visiting the beach. However, they also recoup the cost of driving to the beach, which they would have incurred had the beach been open. Recouping these costs helps to partly offset the lost value. Since consumer surplus is calculated net of expenses such as the cost of driving, it represents what people actually lose when the opportunity to go to the beach is taken away during a beach closure.

In some cases net willingness to pay is expressed as compensating variation rather than consumer surplus. Compensating variation (or "CV") is the theoretically correct measure of willingness to pay and compensatory damages. It represents the amount of money required to make someone as well off as she would have been had the resource injury not occurred. In human-use assessments, consumer surplus is usually used as an approximation for compensating variation because it is simpler to estimate when revealed-preference methods are used. The theoretical drawback of consumer surplus is that it treats each dollar of compensation the same, and fails to account for the fact that money becomes worth less, relative to resource injury, with each additional dollar of compensation paid. The difference between the two value measures is typically small, and the consumer-surplus measure of damages is slightly less than the analogous compensating-variation measure of damages.

3.3 Recreational losses

Recreational activities potentially affected by environmental incidents include fishing, beach use, crabbing, boating, shellfishing, bird watching, picnicking, water skiing, and many others. The most common assessment methods for these activities include travel-cost models, stated-preference models based on choice experiments, and benefit transfer. These approaches are described in Sections 3.1, 3.2 and 3.3. In Section 3.4 we specifically examine methods for the assessment of short-term events, such as oil or chemical spills with impacts lasting a few years or less. In Section 3.5 we examine methods for the assessment of long-term events such as chronic contamination. Many of these methods require obtaining a count of total recreation trips to an affected area, described in Section 3.6. Section 3.7 compiles evidence from past studies on recreation impacts from oil spills and chemical contamination. Section 3.8 discusses scaling methods for recreational losses that do not involve direct monetary valuation.

3.3.1 Travel-cost models

A significant indication of the value of natural resources is the expense people are willing to incur to engage in outdoor recreation. Travel-cost models take advantage of this "revealed preference" information by analyzing the recreation choices people make. In particular, they examine the tradeoff between travel expenses required to reach outdoor recreation sites and attributes of the sites available for the enjoyment of visitors. The quality of natural amenities and the presence or absence of contamination are important resource characteristics that can be evaluated in a travel-cost model.

To develop a travel cost model, researchers implement a survey in which people are asked to report their recreation trips over a given period of time. The survey specifies the type of activity, such as beach use, and identifies the sites of interest. A good example of this type of survey, asking residents of Delaware and New Jersey about their trips to 62 beaches in the summer of 1997, is available on the internet (Parsons, 2004). In addition to behavioral data on recreation trips, surveys typically collect demographic data such as the age, education, and income of survey respondents.

The key to estimating value in a travel-cost model is determining the cost of traveling to a recreation site. For automobile-based travel this is done by adding together direct monetary expenses, such as gasoline and tolls, indirect costs such as per-mile automobile depreciation, and the perceived cost of time spent driving. Estimates of vehicle driving expenses are available from sources such as the American Automobile Association, which estimates a 2008 cost of $0.17 per mile for gasoline and $0.038 per mile for depreciation for an average size car (AAA, 2008). To calculate costs on a per-person basis, vehicle expenses would typically be divided by two or three passengers in a vehicle. The cost of time spent driving is usually estimated as some fraction of an individual's wage rate or hourly income, which may be calculated as annual income divided by 2000 working hours. One-third of the wage rate appears to be the most common estimate of the time-cost of driving (Cesario, 1976; Train, 1998; Moeltner, 2003).

Since survey respondents are often reluctant to report annual income, average income data for the relevant region, obtained from public sources such as the U.S. Census, are a common alternative. Total estimated travel cost in the U.S. typically falls within the range of $0.30 to $0.50 per person per mile applied to the full round-trip distance of a recreation trip.

One objective of travel-cost analysis is to estimate the value of recreation trips. A rigorous derivation of the value of trips involves the area under a demand curve for trips (Parsons, 2004). Intuitively, a travel-cost model determines maximum willingness to pay for trips to a given site based on the maximum distance at which people take trips to the site. Most people who take trips to the site live closer than this maximum distance, and an individual's actual distance to the site represents the individual's actual payment for a trip. Distances are converted to monetary values using the travel-cost calculations described above. The difference between maximum willingness to pay for a trip and the actual trip cost for a given individual is the individual's net value of a trip, also known as consumer surplus.

Another objective of travel-cost analysis is to estimate the value of site attributes. A travel-cost model ascribes value to a given site based on the total value of trips taken to the site. The value of particular site attributes is determined based on the relationship between site value and site attributes, estimated across a variety of sites. This involves the standard statistical concept of regression analysis. If fishing sites with high catch rates are consistently estimated to have high value to anglers, a travel-cost model will ascribe a high value to the catch-rate attribute. When determining the value of site attributes, a travel-cost model controls for the fact that site value is jointly determined by both site attributes and the presence of nearby substitute sites. Important site attributes in addition to catch rates include access to the water and species mix for fishing, the presence of boardwalks, high surf, or natural sand dunes in the case of beach use, wildlife abundance and low hunter density in the case of hunting, etc. For any of these activities, the presence of environmental contamination can be an important factor determining the value of natural resources for outdoor recreation.

An example of the use of travel-cost models in NRD is provided by the assessment of recreational fishing losses on the Clark Fork River in southwestern Montana. Details of the recreational fishing assessment conducted for the state of Montana are described in Morey *et al.* (2002). Trout fishing is popular in southwestern Montana and many trout streams in the area attract anglers from nearby cities such as Butte, Helena and Bozeman, as well as from throughout the United States. However, ecological studies determined that the release of heavy metals from mining waste had caused a significant decline in trout stocks over a 125-mile stretch of Clark Fork River and had completely eliminated trout from one of the river's tributaries. The decline in trout stocks had likely caused a decline in catch rates and a decline in the value of the contaminated streams as a destination for recreational anglers.

To estimate damages, experts for the state of Montana developed a travel-cost model based on 26 major trout-fishing sites in the region. Sites consisted either of individual creeks or larger streams divided into specific segments, and the contaminated streams were among the 26 sites. The attributes used to describe the sites included the cost of traveling to each site for a given individual, the expected catch rate at a site, the size of a site, the presence of a campground at a site, and the suitability of a site for float fishing. Anglers were approached at the sites and asked to participate in a survey that would record their trout fishing activity throughout the region over the course of the season. Residents of Montana who participated in the survey took an average of 6.4 trips per season to southwestern Montana, and nonresident anglers took an average of 1.3 trips per season. The number of trips taken to the contaminated Clark Fork sites was low compared to other sites in southwestern Montana.

The analysis consisted of a RUM model that accounted for the effect of contamination on the total number of trout fishing trips to the region, in addition to the choice of which site to visit within the region. The model estimated an annual willingness to pay for the absence of contamination of \$6.31 per year for residents and \$14.17 per year for nonresidents. Extrapolating to the population of resident and nonresident anglers, total damages were estimated to be \$1.4 million per year in 1992 dollars. An alternative estimate of losses was developed by representatives of Atlantic Richfield Co., one of the responsible parties in the case. Details of that assessment are contained in Desvousges and Waters (1995). One significant area of dispute involved the method used to estimate catch rates, and whether data on the choice of sites by anglers should be combined with site-specific catch data to estimate catch rates, or whether observed catch rates alone generate the statistically correct estimate of catch rates (Morey and Waldman, 1998, 2000; Train *et al*., 2000). Atlantic Richfield agreed to a partial settlement of the Clark Fork claim in 1998 for \$130 million, an amount that addressed the State's recreational fishing claim but also included ecological damages.

Other travel-cost studies conducted for NRD cases include an assessment of losses to recreational fishing conducted by industry experts in the Green Bay case (Desvousges *et al.,* 2000; MacNair and Desvousges, 2007). Among the noteworthy advances in this study was the use of stated-preference data to combine a complex array of species-specific advisory levels into a single advisory index that appeared as a site characteristic in a multiple-site travel-cost model. Also worth noting was the adjustment of losses in calculated using a model of single-day trips to account for losses associated with multiple-day trips. As is common in the literature, Desvousges *et al.* (2000) estimated a model based on data for single-day trips only. They then multiplied losses from the day-trips model by the ratio of total fishing days, including multiple-day trips, to the number of single-day fishing days included in the model. This adjustment assumes that the value of a fishing day is the same regardless of the number of days in a trip, *i.e*., a two-day trip has twice the value of a one-day trip, and so on.

The assumption that a day of recreation is worth the same for both single-day and multiple-day trips is implicit in many NRD assessments. For example, when a change in beach use is estimated based counts of people at the beach, the user counts do not distinguish between people who are engaged in single or multiple-day trips. The resulting estimate of a decline in user days is typically multiplied by a single-day user value (e.g., Chapman and Hanemann, 2001; Byrd *et al.,* 2001). When multiple-day trips have been explicitly modeled in the literature, they have either been combined with single-day trips in the same demand function (Englin *et al.*, 1998) or have been addressed using a separate demand function in a nested model (Shaw and Ozog, 1999; Lupi *et al*., 1998). An evaluation of appropriate methods to account for multiple-day trips in recreation assessments is a worthwhile topic for further investigation.

Following an oil spill in Charleston, South Carolina, losses to recreational shrimping were assessed using a repeated-logit model of site choice and participation (English *et al.,* 2004). The term "participation" typically refers to the choice of how many trips to take in a given year, which is modeled together with the choice of which site to visit on each trip. Data were collected for the model using a survey of people who had purchased a license for recreational shrimping. Like many similar studies (*e.g*., Morey *et al.*, 1993; Parsons *et al.*, 1999), the south Carolina shrimping model estimated the total number of trips people take without accounting for self-selection in the population of license holders, leading to biased estimates of the effect of resource impacts on the value and demand for recreational shrimping. This bias was corrected in English (2008), which extended the participation component of the model to include the choice whether to purchase a license, thus accounting for the way people self-select into the population of license-holders. The revised model also allowed value associated with license fees to be counted in economic estimates of losses and gains.

3.3.2 Stated preference methods

Stated preference models involve the use of survey questions which are specifically designed evaluate particular resource characteristics. In this way they overcome the most important limitation of travel-cost models, namely, the difficulty of identifying which characteristics of a resource people are responding to when they decide where to recreate. For example, a large river may be contaminated with industrial pollutants, while the tributaries to the river may be comparatively clean. Those who choose to go fishing on the tributaries may prefer the cleaner water, or they may prefer smaller streams to large rivers. It can be difficult to determine the value of clean water if there is no direct real-world comparison between the contaminated river and a similar large river that is clean. Stated-preference questions can directly address the issue by asking people how much they would be willing to pay to be able to fish in the contaminated river if it were cleaned up.

Many important topics associated with stated-preference methods were discussed in the context of ecological assessments in Sections 2.7.1 and 2.7.2. These topics include selection of an appropriate payment vehicle and addressing problems of survey design and interpretation including scenario rejection, protest, and hypothetical bias. This section provides additional detail on aspects of stated-preference methods specifically associated with the valuation of recreational use.

As in ecological assessments, stated-preference analysis for the assessment of NRD losses to recreation is likely to involve choice experiments. In recreation valuation, choice experiments typically describe to respondents two or more recreation sites, and ask respondents to select the preferred site. The site alternatives are characterized by particular site attributes, and the levels of the attributes differ across alternatives. For example, a choice-experiment study of fishing might include alternative catch rates and different levels of water quality.

The mechanism for eliciting willingness to pay is described as the "payment vehicle". In recreation valuation, the payment vehicle is often incorporated into the context of recreational use. Common examples include user fees for access to recreation sites and a specified travel distance to reach a recreation site. Morey and Breffle (2006) used boat launch fees as a payment vehicle, and included alternative levels of the fee among the site attributes presented to respondents. The site attributes also included catch rates and various levels of a fish consumption advisory. DeShazo and Fermo (2002) used a fee for visiting archeological sites as a payment vehicle, and explored site attributes such as the likelihood of seeing wildlife and the availability of amenities like a hotel and restaurant. In a contingent valuation study, Kinnell *et al.* (2002) used the fees collected for annual duck-hunting stamps as a payment vehicle, and asked hunters whether they would pay higher fees in order to fund a program that would preserve wetland habitat for ducks.

Many studies have used travel distance to elicit willingness-to-pay values (Lavaca Bay Trustee Council, 1998; Adamowicz *et al.,* 1999; Haener *et al*., 2001; MacNair and Desvousges, 2007). In the choice between hypothetical sites, the number of kilometers or miles to each site is included along with other site characteristics, such as catch rates or availability of campgrounds. When respondents choose distant sites with characteristics more desirable than those of nearby sites, they express a willingness to drive further for characteristics they like. Driving distance can be converted in a monetary value using assumptions about the cost of gasoline and the perceived cost of time spent driving, as in a travel-cost model.

Choices presented in a survey must make sense to respondents and must be consistent with any previous knowledge respondents may have. If this is not the case, surveys may not obtain valid information on respondent preferences due to scenario rejection. In NRD assessment, damages are often assessed for past years, funds for addressing damages will be provided by responsible parties, and the specific circumstances of many sites are well known to the general public. This can make it difficult to develop choice-experiment scenarios, which typically describe future environmental improvements funded by government through taxes or fees. For example, people might be reluctant to agree to higher recreational user fees in order to fund water-quality improvements at a lake, because they may know that companies responsible for contaminating the lake are supposed to be held liable for cleanup and restoration. In this case, stated willingness to pay may understate true willingness to pay due to rejection of any scenario in which public funds are used for cleanup. To avoid issues associated with fees and taxes, another possible scenario might ask anglers to choose between a contaminated site and a clean site, with travel distance as a payment vehicle. This could lead to scenario rejection if major sites in the area are all part of the same contaminated system and people conclude that some sites could not truly be clean while others remain contaminated.

There is often controversy surrounding the use of stated preference methods in environmental valuation, and controversies are most apparent on the topic of hypothetical bias. Section 2.7.1 discussed the examination of hypothetical bias using market goods for which stated values and actual market transactions can be compared. For the assessment of recreational use, the results of stated-preference studies can be compared to the results of revealed-preferences studies to test what is known as “convergent validity”. The comparison to revealed-preference methods is especially useful because revealed-preference estimates are not subject to hypothetical bias. Carson *et al.* (1996) examined 83 studies that valued the same good using both stated and revealed preference methods. For a broad selection of nonmarket goods they found that stated-preference methods on average resulted in smaller value estimates than revealed-preference methods. For recreation values in particular, Carson *et al.* found that stated-preference values were about 75 percent of revealed-preference values.

Only a small amount of research into hypothetical bias has specifically involved choice experiments. As noted earlier, current NRD practice favors attribute-based choice experiments over other stated preference methods. Carlsson and Martinsson (2001) provided an early examination of hypothetical bias using a very simple choice-experiment format. The authors found that choice experiments performed well when compared to actual monetary contributions or market purchases. In a closer examination of hypothetical and actual transactions, List *et al.* (2006) found that the overall choice of whether to make a purchase was subject to hypothetical bias, but that the estimated (marginal) values for particular attributes of the purchased good were not subject to bias. By contrast, MacNair and Desvousges (2007) compared choice experiments to revealed-preference methods and found that respondents overstated their willingness to travel further for attributes they liked, leading to an upward bias in the valuation of choice-experiment attributes.

An important consideration in developing choice experiments is the method for selecting appropriate attribute combinations to be presented in choice scenarios, known as "experimental design". While the attributes to be investigated are selected based on their importance to respondents and their relevance to the research objective, the way attributes are combined and presented in survey questions is based on statistical efficiency. The simplest experimental design involves the random selection of attribute levels. If there are two attributes (e.g., fish consumption advisories and catch rates) with three levels each (e.g., no consumption, one meal per month, or one meal per week and one, two or three fish per hour) then there are 3^2, or nine, possible attribute combinations. For each choice question, one could randomly select two different combinations out of the nine possibilities. For example, one might ask whether a respondent prefers a one-meal-per-week advisory combined with a catch rate of one fish per hour, or a higher catch rate (e.g., three per hour) at the cost of tolerating a more stringent advisory (e.g., no consumption). Choice experiments based on random combinations of attribute levels represent an improvement over revealed preference data, because real-world choices are likely to be characterized by systematic correlation among the levels of some attributes.

Non-random designs can obtain more robust estimates of preference parameters and willingness to pay by ensuring that choice questions do not elicit redundant information. A "full factorial" design would involve presenting respondents (collectively) with every possible choice alternative, where each alternative consists of a unique combination of attribute levels. This approach is often scaled back to a "main effects" design, which uses a subset of attribute combinations sufficient to evaluate the marginal value of any given attribute independent of the level of other attributes. Sometimes the combined effect of a change in two or more attributes differs from the sum of the independent effects of the attributes. A "main effects plus interactions" design evaluates these attribute interactions in addition to the main effects. A larger number of observations is required to evaluate a design that includes interactions. Two-way interactions are often evaluated, but higher-order interactions are unlikely to occur in economic behavior and are usually ignored in experimental design (Holmes and Adamowicz, 2004).

NRD assessments require respondents to state their willingness to pay to have a clean resource rather than a contaminated resource. Attainment of a clean resource is usually presented as the result of a specific policy, such as spill prevention. The most realistic policy option in the case of long-term contamination is cleanup of the contaminated site. Unfortunately, cleanup operations can entail costs that reduce willingness to pay for cleanup. These include the short-term risks to aquatic life from dredging of contaminated sediments, the construction noise associated with dredging operations, or the hazards posed by disposing of contaminated sediments. The theory and practice of NRD suggests that these costs should be additive with other environmental losses in assessing damages. However, stated-preference methods based on willingness to pay for cleanup would evaluate losses net of these costs. This is because costs associated with cleanup reduce the net value of cleanup to the public. Some people may even oppose a plan for contaminant removal, resulting in negative willingness to pay in a stated-preference survey, while placing a positive value on a clean resource absent the drawbacks of a cleanup process. Researchers can attempt to identify respondents who are sensitive to the particular costs associated with cleanup and remove the negative influence of these respondents on total willingness to pay. However, adjustments are likely to be difficult and value estimates in these circumstances are likely to be a lower bound.

Several NRD assessments have used stated-preference studies to evaluate recreational use. Desvousges *et al.* (2000) and Breffle *et al.* (1999) both evaluated fish consumption advisories in the lower Fox River and Green Bay, Wisconsin. These two studies were developed by opposing groups in the Green Bay assessment, and both studies were later published (Morey and Breffle, 2006; MacNair and Desvousges, 2007). Lavaca Bay Trustee Council (1998) describes a study of fish consumption advisories in contaminated waters near Galveston, Texas. The Lavaca Bay study was conducted cooperatively between consultants for the natural resource trustees and the responsible party.

3.3.3 Benefit transfer methods

Benefit transfer for outdoor recreation involves the evaluation of resource changes at a recreation site using previous studies drawn from the economics literature. A valid benefit transfer requires selecting previous studies that have examined the same recreational activity in a similar context. The similarities should include characteristics of the resource, demographics of the user population, and the availability of nearby substitute sites. The resource valuation literature distinguishes between value transfer and function transfer. Value transfer involves multiplying recreation trips at the site of interest by an appropriate per-trip value drawn from the literature. Function transfer involves developing a model applicable to the site of interest by adapting models previously applied to other sites. We focus on value transfer and the use of per-trip values, since benefit-function transfer generally involves the same concepts described for valuation models in Section 3.1.1. Many authors have examined benefit transfer in detail, including the criteria for selecting literature studies, the validity of value and function transfer, and other topics. Interested readers should refer to Rosenberger and Loomis (2001), Champ *et al.* (2003), and Navrud and Ready (2007).

This section describes value-transfer methods applicable to recreational losses from spills or chronic contamination. In the case of short-term impacts from a spill event, it is often possible to estimate the total change in recreation trips to an affected area. Pre-spill or post-spill data can be used as an estimate of baseline activity, and actual activity observed during the period of impact can be compared to the baseline estimates. The difference between baseline trips and actual trips provides an estimate of trips diverted from the affected area, and the number of diverted trips can be multiplied by an average per-trip value to determine spill-related losses. In the case of long-term contamination, it may be difficult to estimate the number of diverted trips because the baseline level of activity is unknown. In this case it may be preferable to develop a lower-bound estimate of losses based on the number of trips taken under degraded conditions multiplied by an appropriate unit loss for degraded trips. This method works well when a site is unique and demand for the site under degraded conditions remains high. When the number of diverted trips is difficult to estimate but is likely to be high, lost value may be assessed using an appropriate per-trip loss multiplied by all trips taken throughout the surrounding region. These three approaches to benefit transfer are described next.

3.3.3.1 Diverted trips multiplied by average consumer surplus per trip

The recreation demand literature provides numerous estimates of the value of recreation trips. Reviews of literature values can be found in Freeman (1995), Rosenberger and Loomis (2001), Loomis (2005), and elsewhere. The most common value reported in the literature is average consumer surplus per trip, which is usually obtained from a travel-cost model. Average consumer surplus per trip is calculated by simulating the closure of a recreation site and then dividing the resulting loss in value by the number of trips initially taken to the site.

If there is a linear relationship between recreation trips at a site and consumer surplus from trips, then average consumer surplus per trip is a useful value for analyzing resource changes. A linear relationship is the basic assumption in most benefit transfer applications using per-trip values (e.g., Rosenberger and Loomis, 2001). The method can be expressed as:

$$\Delta CS_2 = \Delta T_2 \frac{CS_1}{T_1}. \qquad (3.1)$$

The term CS refers to consumer surplus, T refers to total trips taken to a site, and the subscripts 1 and 2 refer to specific recreation sites. It is assumed that similarities between the two sites are sufficient to support a valid benefit transfer. The number of trips diverted from site 2 (ΔT_2) attributable to a resource change such as an oil spill is multiplied by average consumer surplus per trip at site 1 (CS_1/T_1) to estimate the consumer surplus loss at site 2 (ΔCS_2).

The assumption of a linear relationship between trips and consumer surplus must be evaluated in any given situation. Since the per-trip value in equation (3.1) is calculated based on a complete loss of access at site 1, the most appropriate transfer would involve a complete loss of access at site 2. In many cases a spill incident leads to only a partial decline in use, and in this situation an adjustment to equation (3.1) may be appropriate. In the assessment for the American Trader spill in Southern California (Chapman and Hanemann, 2001) diverted trips were multiplied by a per-trip value drawn from a study of Florida beaches. However, losses were also ascribed to trips that took place under degraded conditions ($T_2 - \Delta T_2$). Experts in the case decided that a 20-percent per-trip loss ($0.2 \times CS_1 / T_1$) would be appropriate for degraded trips. Although this calculation was not derived from any specific theoretical or empirical evidence, it is consistent with many valuation models that estimate a higher level of losses for a partial decline in trips than would be predicted by the linear relationship in equation (3.1).

The benefit transfer analysis in the American Trader case relied on the selection of a specific published study to estimate per-trip values. The broad sandy beaches of Florida and California are quite similar, but in many instances it can be more difficult to select a single study that appropriately captures the attributes of an affected site. Selecting a good study for use in benefit transfer is often difficult because outdoor resources may have numerous attributes which are not explicitly described in published studies. In these instances, it may be preferable to use an average per-trip value calculated from multiple studies. There have been numerous attempts to compile valuation studies and present average or meta-analysis values for selected recreational activities, and recent examples include Rosenberger and Loomis (2001) and Loomis (2005). An important justification for the use of average values is the significance of unobserved factors in determining the value of recreation sites. If one assumes there is some distribution of unobserved factors that has an important influence on the value of a given site, the influence of unobserved factors on per-trip value at an affected site represents a single draw from this distribution. The best estimate of the value of this draw is not another draw from the distribution of unobserved factors, as provided by a single literature study, but rather the mean of the distribution, as provided by an average over numerous studies. The role of unobserved factors in determining individual choice has received increasing attention over the last decade, and the reporting limitations associated with the use of previous literature studies augment the issue in benefit transfer analysis. The use of average values in damage assessment is illustrated by the Athos case (Athos, 2007), which examined losses to fishing, hunting, and boating following an oil spill on the Delaware River.

3.3.3.2 Trips to a degraded site multiplied by WTP_{site}

Often a change in recreation trips attributable to resource impacts is difficult to measure, as when contamination has persisted at a site for many years. In this case benefit transfer methods may be easier to apply using a variant of per-trip value obtained from the estimated coefficients in a travel-cost or stated-preference model. The relevant per-trip value is $WTP_{site} = \beta_{\text{contamination}} / \beta_{\text{price}}$. The numerator is the model coefficient associated with the presence of contamination. The presence of contamination could either be a site attribute in a travel-cost model or an attribute of choice alternatives in a stated-preference survey. The denominator is the coefficient on travel cost or price. These coefficients would typically be reported in literature studies that model the recreation impacts of resource contamination. WTP_{site} represents the consumer surplus loss associated with a recreation trip taken to a site under degraded conditions.

As demonstrated by Morey (1994), one can obtain a lower-bound estimate of losses at an affected site by multiplying WTP_{site} by the number of trips taken to the site in its degraded state. The result is a lower bound because trips to the degraded site represent a subset of all affected trips. Losses are also incurred when people take trips to alternative sites to avoid contamination, or take fewer recreation trips and instead engage in alternative activities. The situation is represented graphically in Figure 3.1.

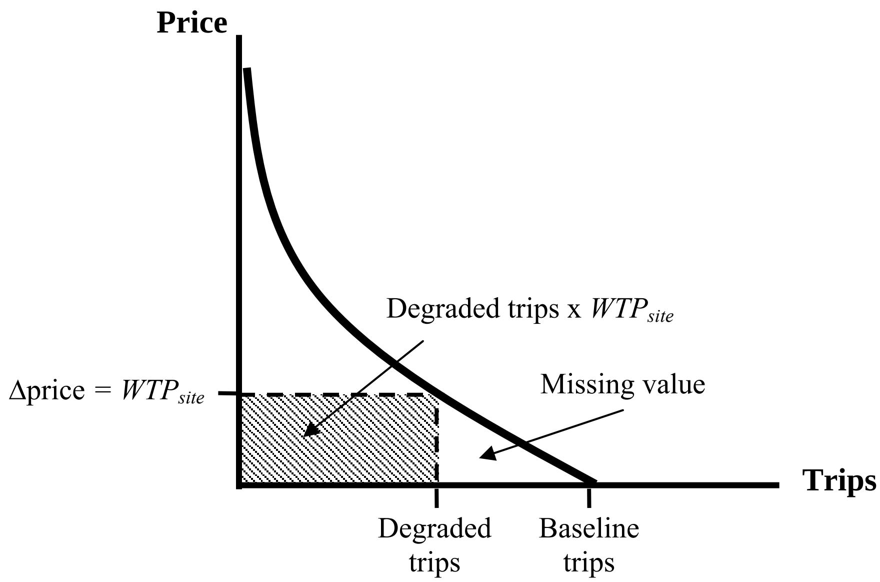

Figure 3.1. Lost value from trips to a degraded site

Figure 3.1 shows the demand curve for trips to a recreation site. It may be viewed as a market-level demand curve, or as an individual-level demand curve for a representative agent with average preferences. The demand for trips to the site is displayed on the horizontal axis and the price of a trip is displayed on the vertical axis. The presence of contamination at a site results in a per-trip loss with value WTP_{site}, which is equivalent to an increase in price for access to the site. Given the presence of contamination and loss in per-trip value, the demand for trips at the site declines from its baseline level. The actual number of trips observed in the degraded site is labeled "degraded trips". The change in price and decline in demand are associated with a loss in value that can measured by the decline in total area under the demand curve. The shaded area gives a lower-bound estimate of the loss, which is calculated by multiplying WTP_{site} by the number of degraded trips. This approach to benefit transfer understates the total loss because it does not include the missing value depicted under the lower right portion of the demand curve.

When a site is unique and close substitutes are not available, the demand curve for trips would be steep and the "missing value" in Figure 3.1 would be small. In this case the use of degraded trips multiplied by WTP_{site} is likely to be the best benefit-transfer approach. This may be the case even if the decline in trips to the site can be estimated and the method in described in Section 3.3.3.1 could be applied. This is because literature values for average consumer surplus per trip may not sufficiently account for the lack of substitute sites, and the small number of diverted trips, when a site offers unique resource amenities.

3.3.3.3 Total trips to the region multiplied by WTP_{region}

If site substitution is believed to be significant but the number of diverted trips is difficult to estimate, neither the first nor the second method described above is likely to be satisfactory. However, a third method may still be applicable. Some literature studies report recreational losses from contamination using a per-trip loss for trips throughout a region. Specifically, WTP_{region} is calculated as the value lost due to contamination at a site or group of sites divided by total recreation trips to all sites in the surrounding study area (e.g., Jakus *et al*., 1998, 2002; Parsons *et al*., 1999). This type of per-trip value is usually derived from a travel cost model that analyzes the demand for recreation trips to multiple sites. Because multiple-site models account for losses from site substitution in addition to losses associated with degraded trips, WTP_{region} accounts for a greater portion of total losses than WTP_{site}. In a benefit transfer exercise, losses due to contamination at a particular site would be calculated as WTP_{region} multiplied by total trips taken to the relevant region.

Two major difficulties must be addressed in the application of regional per-trip values. The first involves defining the relevant region. When examining impacts at particular sites, researchers typically construct a model that includes all reasonable substitute sites. A similar standard could be applied in a benefit transfer, but without undertaking a survey of recreation behavior the relevant region would be difficult to identify. Jakus *et al.* (2002) estimated potential losses to recreational anglers from a proposed mercury advisory in the Maryland portions of Chesapeake Bay. They derived WTP_{region} from several studies of contaminant-based fish consumption advisories. The relevant region for a benefit transfer could have been defined to include all sites in the Maryland or Virginia portions of the Bay, and could even include freshwater sites. The transferred value was applied to trips at a subset of these sites, namely, saltwater sites in Maryland. The result was considered to be a lower-bound estimate of losses from mercury contamination.

Another difficulty involves the extent of contamination within the defined region. A study examining one contaminated site among multiple clean sites would generate a low regional per-trip loss. If this value were then applied to a region in which most of the sites are contaminated, the result could be a significant underestimate of damages. The recreational assessment for the Kalamazoo River (Stratus Consulting, 2005) converted WTP_{region} to WTP_{site} based on the proportion of sites in the region that were subject to contaminant advisories. Specifically, the analysis assumed that if p is the proportion of contaminated sites, then $WTP_{site} = WTP_{region} / p$. A similar type of adjustment could be used to adapt estimates of WTP_{region} from one region to another. For example, if subscripts 1 and 2 represent the respective regions, then $WTP_{region2} = WTP_{region1} \times (p_2 / p_1)$. The numerical proportion of contaminated sites is a crude measure of the extent of impacts in a region, and it may be useful to develop more refined adjustments that also account for the size or popularity of sites.

3.3.4 Assessing losses from short-term events

While the chronic release of contaminants can cause impacts to recreational activities lasting for decades, many spill events lead to acute impacts over a shorter period of time. We will use "short-term events" to refer to incidents whose recreational-use impacts last from several weeks up to one or two years. When assessing losses from short-term events, it is usually possible to obtain information on both degraded and baseline conditions. This is because pre-spill conditions are usually recent enough to be easily recalled by respondents to a survey or examined using data from recreation sites such as state or local parks. The researcher can also wait until spill effects fully subside and observe recreational activity following the return to baseline conditions. A comparison of degraded and baseline conditions permits a convenient measure of the severity of recreation impacts, namely, the change in the number of recreation trips to the site of impact. Changes in the demand for trips can be translated into lost value using commonly accepted methods such as benefit transfer or travel-cost models.

There are three common approaches to obtaining information on changes in trip demand. First, one can rely on recreators' self-reported losses. This involves implementing a survey to directly ask recreation participants whether and to what extent they took fewer trips to the impacted site. Second, one can model historical trends in recreational use as a function of weather and other variables. Data are obtained from recreation sites that routinely track visitation levels and predictions of baseline activity are compared to actual activity during the period of impact. Finally, one can use a control site as a reference. This would typically involve measuring recreational use simultaneously at the spill site and a control site during and after the period of impact, and conducting a comparison of relative use under both degraded and post-recovery baseline conditions.

These three approaches are described below. Each approach leads to an estimate of the percentage decline in recreation trips at an impacted site. To determine total lost trips, the percentage decline must be extrapolated to a count of total trips throughout the area of impact. Methods for counting total trips are described in Section 3.3.6.

3.3.4.1 Survey-based methods

The most direct method for determining the response of recreators to a spill is to interview the recreators. Interviews may be conducted by telephone or in person at impacted recreation sites. Telephone surveys are useful whenever the target population, such as users of a particular site, is reachable through an available list of contact numbers. For example, a list of those who hold a fishing license in a particular county may serve as a reasonable contact list for a telephone survey assessing fishing at a local site. Activities available to the general population, such as beach use, are often assessed using on-site surveys, also called "intercept" surveys. Mail surveys are more expensive and are usually not required because the types of questions involved in estimating affected trips are simple enough to be conveyed orally. However, the continuing decline in response rates for telephone surveys may make mail surveys more feasible in some cases. Assessments that have relied on self-reported losses obtained from on-site surveys include the Athos/Delaware River oil spill assessment (Athos, 2007).

1. *Developing a recreational use survey*

Many survey-based studies require the intensive development of detailed questions for a comprehensive survey instrument. Even a relatively simple analysis may require considerable effort in designing a survey if a case is prepared for litigation. Good sources for this type of survey methodology can be found in books on survey research such as Tourangeau *et al.* (2000) or Dillman (2006). However, in many cases the data and methods required for an evaluation of lost recreational use are simple enough that data collection is relatively straightforward. This section presents an overview of one common survey strategy for evaluating self-reported losses. The example involves an on-site survey, but the questions can easily be adapted to apply to a telephone survey (see, for example, the hunting survey for the Athos oil spill assessment).

Table 3.1. On-site Survey Part A – Information on respondent and intercepted trip

1. Survey #____; Date____; Time____; Location____; Interviewer Initials____.
2. Introduction: Hello, I'm [name] and I'm conducting a survey on behalf of the state of Delaware regarding recreational activity on Delaware Bay…
3. Activity (circle one): Swimming Shore fishing Birding Other_______.
4. Trip duration____; Number of people in group____.
5. Respondent's trip origin (e.g. zip code of residence) ____.
6. Percent of respondent's trips to this location typically taken on weekends____.

An overview of typical questions used in an on-site survey is provided in Tables 3.1 through 3.3. The questions are adapted from three different oil-spill assessments conducted in the State of Delaware over a period of several years. Table 3.1 shows the first part of the survey, which records information about the respondent and the trip the respondent is taking at the time of the interview. Questions 1 and 2 involve introductions and recordkeeping and are included for completeness. Question 3 records the type of activity the respondent is undertaking on the intercepted trip. Subsequent questions refer only to this activity even if the respondent takes part in other activities. This allows activities to be analyzed separately, which may be important if different per-trip values are assigned to different activities, or if impacts and recovery follow different patterns for different activities. Question 4 records the duration of the respondent's trip and the total number of people in the group (including the respondent), which are important for assigning sampling weights to each observation, described further below. Question 5 records the point of origin for the current trip, which is usually the respondent's place of residence. This is typically recorded as a zip code and is included so that researchers may develop a simple travel cost model to assist with valuation of trips. Question 6 records the proportion of trips the respondent takes on weekends versus weekdays at the intercept site. This is discussed further below in the context of assigning sampling weights.

Table 3.2. On-site Survey Part B – Recreation activity following the spill

1. Before today, did you go [activity] on Delaware Bay during April or May?
   Yes No

   [If yes]

   __________ Not counting today, how many days did you go [activity] on Delaware Bay during the four-week period from April 22nd through yesterday (Saturday, April 22nd through Friday, May 19th)?

2. There was an oil spill on Delaware Bay in late April. Did you know about the spill?
   Yes No [If no, quit]

   [If yes]

Has the spill affected your [activity] trips to Delaware Bay in any way, such as how many trips you have taken or where you took trips? Yes No [If no, quit]

[If yes]

Fewer trips:

__________ How many fewer [activity] trips if any did you take to Delaware Bay during the four-week period from April 22nd through yesterday (Saturday, April 22nd through Friday, May 19th) because of the spill?

Changed locations:

__________ How many times if any did you change the location of your [activity] trips on the Delaware Bay during the four-week period from April 22nd through yesterday (Saturday, April 22nd through Friday, May 19th) because of the spill?

Table 3.2 presents questions that elicit information on the respondent's recreational activity in the period following the spill. In the first question the respondent is asked to recall the number of recreation trips he or she took to the impacted site in the four-week period since the spill. Words appearing in brackets represent instructions to the interviewer, and "activity" refers to the respondent's answer to question 3 in the previous section. The first part of question 1 introduces the theme of the question and allows the respondent to begin the process of recalling his or her recent activity. The second part of the question addresses the post-spill period more specifically. Question 2 begins by asking whether the respondent had heard of the spill. If not, the interview is ended, because the respondent could not have changed recreation behavior in response to the spill.

If the respondent has heard of the spill, the next part of question 2 elicits information on any change in recreation behavior due to the spill. The typical changes in behavior include taking fewer recreation trips or changing the location where trips are taken. The first response is usually referred to as lost trips and the second response is usually called substitute trips. It is usually easier to obtain concrete responses if the two types of behavioral changes are specifically introduced. Vague questions about changes in recreation activity often lead to less informative responses. When referring to a change in the location of trips, it is important to avoid any specific reference to destinations outside the "impact area" where oiling occurred, because respondents are unlikely to be familiar with such details. These details are not relevant to the survey because avoidance of an area perceived to be affected by oil represents a loss attributable to the spill whether or not the area was actually affected.

Table 3.3. On-site Survey Part C – Current conditions

1. Has your use of Delaware Bay returned to normal, in terms of [the effects they mentioned above]?
   Yes No

   [If no] Explain____.

2. Has the quality of areas you go to for [activity] on Delaware Bay returned to normal since the spill, in your opinion? Yes No

[If no] Explain____.

Thank You!

Table 3.3 addresses resource conditions at the time of the survey as perceived by the respondent. Question 1 evaluates recreation behavior explicitly by eliciting any ongoing effects of the kind reported in the previous section, namely, lost or substitute trips. Question 2 addresses spill effects more generally by asking whether the quality of the resource has returned to normal. Ongoing losses would likely be associated with a change in behavior for some individuals (question 1), but other individuals may have less elastic demand. Therefore the perception that conditions have not returned to normal (question 2) may also be an important indication that additional assessment surveys will be required at a later date.

2. Calculating statistical weights for on-site surveys

The analysis of survey data collected through on-site surveys requires the development of sampling weights to eliminate bias. Sampling weights may not be required for a telephone survey if it is conducted using a simple random sample of the target population. However, low response rates can be a problem in telephone surveys and relatively complex weighting schemes may be developed to mitigate bias from nonresponse (see Zimowski *et al.*, 1997). Once the appropriate weights are developed, they are applied to the data in a multiplicative fashion, usually to estimate a sample average. For example, if a recreator reporting one lost trip is weighted 0.25 and another reporting two lost trips is weighted 0.75, the average number of lost trips per person is 1.75 ($0.25 \times 1 + 0.75 \times 2 = 1.75$). This differs from the unweighted average, which would be 1.5 trips per person.

The key to understanding sampling weights for an on-site survey is to recognize that some individuals are more likely to be interviewed than others. For example, someone who spends every day at the beach is more likely to be interviewed in an on-site survey of beachgoers than someone who rarely goes to the beach. In the language of sampling, interviews are called observations and the likelihood of being interviewed is known as the probability of selection. To develop a sample that is representative of the target population (e.g., beachgoers), selection probabilities must be exactly offset by sampling weights which are applied to each observation.

Sampling weights for the survey of Delaware Bay described above can be developed based on the information in Tables 3.1 and 3.2. An on-site survey is usually conducted according to a "roving" design in which interviewers visit a series of sites, interviewing anyone present at each site before moving on to the next one. For the purpose of developing appropriate sampling weights, the essential information to be drawn from the surveys is trip duration, trip frequency, the proportion of the respondent's trips taken on weekends, and the number of people in the respondent's group. If trip duration is d, then the weight associated with trip duration is $1/d$. This is because a larger d is associated with a greater probability of selection, causing those who take long trips to be overrepresented in the sample. Weighting by $1/d$ offsets this effect. Likewise for trip frequency, if n is the number of trips the respondents takes per month, the appropriate contribution to the sampling weights is $1/n$. For weighting purposes it is important to estimate trip frequency at the time of the survey, since this determines the probability of selection. For those whose behavior is no longer (or never was) affected by the spill at the time of the survey, the sum of actual and lost trips in the previous four-week period is a reasonable estimate of n. The survey is conducted throughout the entire Delaware Bay, so substitute trips are a component of actual trips and are not added back in to calculate n. For those whose behavior continues to be affected by the spill, lost trips are excluded and actual trips during the previous four weeks could be used as an estimate of n.

The proportion of trips taken on weekends is used to further refine the weights. Consider the case where one person takes seven trips per week and another person takes two trips per week. If the first person takes one trip every day of the week and the second person takes one trip every weekend day, a survey conducted on the weekend would intercept either individual with equal probability. The trip-frequency weight of $1/n$ would incorrectly give greater weight to the person who only takes two trips per week. Another weight in addition to $1/n$ is required to offset this effect. If p is the proportion of trips the respondent takes on weekends, the weight $1/p$ would adjust the weights for a survey conducted on the weekend. For a survey conducted on a weekday, the additional sampling weight would be $1/(1-p)$.

Finally, the number of people accompanying the respondent on the intercepted trip should also be factored into the sampling weights. Specifically, when approaching groups it should be determined whether people in the group typically take trips together. If so, one person in the group may be interviewed and the resulting observation should be weighted by g, the number of people in the group. If an intercepted group does not typically take trips together, each person should be interviewed separately. Each of the separate interviews is not weighted for group size (or, equivalently, is weighted by $g = 1$ for the individual interview). If it is not feasible to interview each person in a group, then one person should be selected at random and the interview should be weighted by size of the full group. In this case the randomly selected respondent is viewed as representative of the full group. This approach should result in unbiased estimates of the number of affected trips, though if standard errors are calculated they will be biased downward.

The final sampling weights are the product of weights for each of the individual factors that influence selection probabilities. For all four factors described above, the final sampling weights are $w = (1/d) \times (1/n) \times (1/p) \times g$. These weights would apply to the roving intercept survey of shoreline sites along Delaware Bay. Other sampling procedures would lead to different sampling weights. For example, if interviewers were stationed at boat ramps and tasked with interviewing people as they returned from their trips, the duration of a person's trip would not influence the probability of being interviewed. The appropriate weights in this case would exclude the factor $1/d$ and would become $w = (1/n) \times (1/p) \times g$. It may still be important to collect information on trip duration, for example, to assist in converting counts obtained by helicopter into an estimate of total recreation trips throughout an extensive area (see Section 3.3.6). The next section describes how to combine information from multiple interviews to determine the overall impact on recreation trips, including a specific illustration of the use of sampling weights.

3. *Calculating the impact on recreation trips*

Data from on-site or telephone interviews must be analyzed to determine the number of lost and substitute trips as a percent of total recreation trips. The percentage impacts can then be applied to an estimate of recreation trips throughout the impacted area (Section 3.3.6) to estimate total losses. The number of lost trips as a percentage of total baseline trips is given by:

$$\%\ \ Lost = \frac{\sum_i w_i Lost_i}{\sum_i w_i Actual_i + \sum_i w_i Lost_i}. \tag{3.2}$$

The weight for individual observation i is w_i, and lost and actual trips for individual i are $Lost_i$ and $Actual_i$, respectively. The denominator represents baseline trips, which include trips that were taken following the spill (actual trips) plus the additional trips that would have been taken in the absence of the spill (lost trips). Following the example of a survey conducted throughout Delaware Bay, substitute trips to alternative locations within the Bay are a component of actual trips to the Bay and are not added into the denominator of equation (3.2). A similar calculation describes substitute trips as a percent of baseline trips:

$$\%\ \ Substitute = \frac{\sum_i w_i Substitute_i}{\sum_i w_i Actual_i + \sum_i w_i Lost_i}. \tag{3.3}$$

The distinction between lost and substitute trips often depends on the context of the survey and the wording of the survey instrument. In the case of the Delaware Bay survey, respondents were asked whether they took fewer trips to Delaware Bay and whether they changed the location of their trips within Delaware Bay. It is convenient to identify these two responses as lost and substitute trips, but the use of these terms may differ in other contexts. For example, some of the trips lost to Delaware Bay could have been taken to alternative recreation destinations outside the Bay and could also be called substitute trips. In practice the specific distinction is not important as long as the various quantities are treated consistently when quantifying total effects.

A distinction between lost and substitute trips is sometimes made when applying trip values in a benefit transfer. However, such an approach is difficult to justify. Average trip values obtained from the literature are typically calculated based on a site closure. When a site is closed, the total number of affected trips (T_1 in equation 3.1) includes both trips taken to alternative sites and trips forgone in exchange for alternative activities. Because no distinction is made between lost and substitute trips in the original study, there is no need to make any distinction when transferring the derived values to a new location.

One drawback of using self-reported losses is the potential for inaccuracies in respondents' recollection of their recreation activities. For example, there is some evidence that respondents overestimate the level of their past recreational activity (e.g., Chu *et al.*, 1992). The use of surveys to calculate a percentage impact may reduce any influence of recall bias on assessed losses, since recall effects would cancel out if both actual and affected trips are overstated in equal proportion. In an assessment of losses to recreational hunting following an oil spill on the

Delaware River, the results of a lost-trips survey were corroborated by evidence from external sources (Athos, 2006, 2007). Specifically, the damage estimate based on self-reported losses was compared to an alternative estimate derived from historical records maintained by the U.S. Fish and Wildlife Service. Though a year-to-year comparison of historical figures was not considered statistically precise enough to serve as the primary basis for the assessment, this alternative calculation suggested an estimate of losses very similar in magnitude to the results of the lost-trips survey.

3.3.4.2 Time-series visitation models

Assessing losses from a spill requires a comparison between actual and baseline conditions. When relying on self-reported losses, the baseline level of activity is calculated by adding up actual and lost trips to estimate baseline trips. An alternative approach is to predict baseline trips using a model that combines historical data on visitation at the impacted site with information on weather conditions to predict the expected level of visitation absent the spill. Historical trends at the impacted site serve as a good reference for comparison to actual conditions as long as weather and any other confounding variables can be adequately addressed. Cases that have used this approach include the assessment of lost beach use following the American Trader incident in southern California (Chapman and Hanemann, 2001) and the assessment of lost beach use and boating following the Chalk Point oil spill in Maryland (Byrd *et al*., 2001).

The essential steps to assessing losses using visitation models are: (1) obtain daily visitation data for several years prior to the spill and also for the period of spill impact; (2) estimate a model that fits visitation in previous years to explanatory variables such as temperature and precipitation; (3) estimate baseline activity for the period of the spill using data on weather conditions during the spill-impact period; and (4) estimate diverted trips by comparing model predictions to actual visitation during the spill-impact period.

Many state and local parks routinely collect data on daily visitation. However, issues sometimes arise when evaluating the quality of the data (e.g., the discussion in Chapman and Hanemann, 2001). Hourly data from weather stations throughout the U.S. are available online from the National Climatic Data Center of the National Weather Service. Variables that may be useful for predicting outdoor recreation activity include rainfall, cloud cover, the presence of fog, wind speed, and temperature. A model of daily activity would also include variables identifying weekdays, weekend days, and holidays. Visitation models use regression analysis to predict aggregate visitation, and the equation for the total number of trips usually takes the form

$$\ln(trips_t) = \beta_0 + \beta_1 weather_t + \beta_2 weekend_t + \beta_x x_t \,. \qquad (3.4)$$

The natural log of trips observed at a site on day t is a linear function of weather variables such as temperature and precipitation, a binary identifier for weekend days versus weekdays, and other independent variables x_t as deemed appropriate. The natural log on the left-hand side ensures that explanatory variables such as temperature cause an identical percentage change in trips regardless of the absolute number of trips. For example, the difference between a cool day and hot day would be expected to cause a large difference in the absolute number of trips observed on a weekend day. However, such a large difference is unlikely to be observed when comparing hot and cool days during the week. It may be therefore reasonable to assume that the percentage change in attendance attributable to a hot day would be the same on weekdays and weekend days despite differences in absolute attendance levels. Econometric estimation could employ a variety of assumptions about the appropriate error distribution that fits predicted trips in equation (3.4) to observed visitation data, resulting in an ordinary least squares regression or other estimators.

A simple but effective approach to specifying equation (3.4) involves summarizing weather variables into a single binary indicator for fair-weather days and foul-weather days. This requires specifying threshold levels for precipitation, wind and other weather data that are combined to define a foul weather day. For example, in Byrd *et al.* (2001) a foul-weather day was defined by the occurrence of one or more of the following indicators: (1) average wind speed greater than 12 knots; (2) rain greater than 5 percent of the day; or (3) fog greater than 20 percent of the day. These thresholds were chosen by visual inspection of weather data and attendance data to identify factors likely to produce a good fit. If the response of recreation demand to weather variables is kinked rather than continuous, or if the effect of multiple foul-weather factors is not cumulative but rather the presence any single foul-weather factor significantly discourages visitation, this binary repackaging of weather variables is likely to generate good predictions. The Byrd *et al.* (2001) model also included a binary weekend-weekday variable and an interaction between weekends and good weather. Estimation employed ordinary least squares to fit the regression equation.

In visitation models estimated for the American Trader spill, data on weather characteristics entered directly into the estimating equation (Chapman and Hanemann, 2001). Maximum and minimum daily temperatures and the presence of rain and were among the explanatory variables. There were also dummy variables for weekends and holidays, as well as linear and nonlinear time trends. Different models were estimated by experts for the government and experts for the responsible party. All models estimated visitation at several beaches using a system of equations with a semi-log form as in equation (3.4) but with alternative error structures for the regression equation. An area of dispute between the two parties involved the use of lagged visitation data among the explanatory variables. Chapman and Hanemann (2001) defended the use of lagged variables, citing their extensive use in economic forecasting and the importance of improved fit when prediction is the primary objective. Predictions of baseline activity were considerably higher when lagged variables were included in the model.

The relationship between weather variables and recreational activity has been investigated outside the arena of NRD. For example, it has received considerable attention in the analysis of impacts from climate change (e.g., Mieczkowski, 1985; Scott *et al*., 2004; Amelung *et al.*, 2007). An analysis that examines the validity of time-series visitation model in NRD based on evidence from elsewhere in the literature would be a worthwhile topic for further investigation.

3.3.4.3 Control-site models

Models described in the previous section depend on the ability of the researcher to correctly define the relationship between aggregate site visitation and selected explanatory variables. There are two difficulties with this approach. One is the potential omission of relevant explanatory variables, such as school vacation schedules or forecasted as opposed to actual weather conditions. The other is the use of functional forms that are unlikely to correspond precisely to relationships in the data. Neither the continuous relationship between temperature and recreation trips in the American Trader models nor the binary fair-foul summary of weather variables in the Chalk Point study is likely to be more than a rough approximation. The use of a control-site model can overcome these difficulties by capturing both observed and unobserved variables that influence recreational activity without assuming a specific functional relationship.

A control-site model is based on counts of people engaged in shoreline recreation throughout the spill-impact area, as well as counts conducted in control areas that are not impacted by the spill. In one recent assessment, visual counts were obtained from the air during helicopter overflights covering approximately 90 miles of shoreline. The shoreline was divided into 24 segments for the purpose of recording the recreator counts. A series of 10 overflights were conducted, five during the month following the spill and five during the same period a year later. While numerous factors can cause recreational use to vary from one day to the next, it was assumed that most of these factors would affect use in a similar way throughout the assessment area. For example, warm days would lead to higher levels of use in all 24 locations. It was assumed that only the oil spill would affect certain locations differently than others in a systematic way. Specifically, there were 21 locations where oiling occurred and 3 locations that were oil free. It was expected that recreational activity in the oiled locations relative to activity in oil-free locations would be lower in the year of the spill than in the subsequent year, when spill effects had subsided.

The effect of the spill was quantified based on the following relationship:

$$use_{ijk} = \exp(day_i + site_j + hour_k + spill_{ij} + substitution_{ij}). \qquad (3.5)$$

The dependent variable use_{ijk} represents the number of recreators counted during overflight i at site j in hour k. The explanatory variables are all binary dummies defined as:

> day_i = day-specific effects for overflight day i;
> $site_j$ = site-specific effects for site j;
> $hour_k$ = time-specific effects for hour k;
> $spill_{ij}$ = spill effects, where i is among the spill-year overflight days and j is among the spill-impacted sites;
> $substitution_{ij}$ = substitution effects, where i is among the spill-year overflight days and j is among the sites identified as likely substitutes for spill-impacted areas.

The effects associated with the *day* variable included the influence of weather, weekend-weekday use, school vacations, and so on, for each day when an overflight was conducted. Recreation trips vary by geographic *site* because some areas have state parks with wide sandy beaches and other areas have narrow or rocky beaches or are adjacent to private land. The start time for overflights varied and each flight took several hours. Controlling for changes in use throughout the day, the *hour* variable generally showed increasing use from the morning hours through mid-afternoon with use tapering off toward evening. For the five overflights conducted during the spill year, the variable *spill* entered the model in all 21 sites where oiling occurred.

The *substitute* variable captured potential effects of substitution that were anticipated during the spill year in one of the oil-free sites where a state park was located. The other two oil-free sites included only town beaches open to primarily to local residents and these two sites together served as the model's control area.

Equation (3.5) was estimated using a Poisson regression, a specification useful for recreation data because it accounts for the fact that observations on recreation trips are nonnegative integers. Once the model parameters were estimated, predictions of baseline use during the spill year were obtained by removing the spill dummy. These predictions applied specifically to the five days when overflights were conducted during the period of the spill. The model was adjusted to predict activity throughout the period of the spill using time-series data that were available for several sites within the spill area. Specifically, the *day* variable was adjusted so that predicted activity at the relevant sites throughout the impact period corresponded to the available time-series data.

A similar comparison between recreation counts in the period following a spill and additional counts taken one year later was used in the assessment for the *Berman* case (Tetra Tech, 2006). The Berman assessment did not use a control site and therefore relied on the assumption that differences in weather were small enough to ignore across various days when counts were conducted and across two separate years. The *Berman* spill occurred in January in Puerto Rice, where this simplifying assumption may be reasonable.

3.3.5 Obtaining a count of total recreation trips

Estimating the value of resources for outdoor recreation often requires an estimate of the number of recreation trips taken to a particular area. A decline in the number of trips to a recreation site following a spill, for example, can be multiplied by a per-trip value to obtain an estimate of spill-related losses. Often recreation impacts are measured as a percentage decline in trips, which is combined with a count of total trips to estimate total impacts.

The most straightforward approach to obtaining a total count of recreation trips is to observe all people entering a particular location over the course of several days. The days when observations are made would be selected so that they are representative of the period of interest. This approach is not always completely accurate, because in some instances it may be difficult to keep track of people who enter and exit a location several times during the day. However, the primary drawback of this approach is the high level of effort required to observe arrivals over the course of several days. The level of effort is higher if the area of interest has numerous points of access.

The method described below is more complicated than a simple count of arrivals, but requires considerably less effort. We follow procedures outlined in Deacon and Kolstad (2000), which in our experience represent the most common approach to counting recreation trips over a large area. The approach involves instantaneous counts of the number of people using an area taken at various times during the day. From these counts it is possible to determine the total number of person-hours spent at the area throughout the day. At the same time "roving" intercept interviews are conducted to determine the average number of hours per trip. Analysis of the interview responses must account for the fact that people who spend more time at the site have a higher probability of being interviewed. Dividing total hours by the average number of hours per trip gives an estimate of total trips per day. Separate estimates of daily recreational activity are typically developed for weekdays and weekend days, and for different times throughout the season.

Specifically, the total number of person-hours at a site is represented by the formula

$$H = \sum_t C_t L_t \,. \qquad (3.6)$$

Total person-hours H spent recreating in a given day is the sum of person-hours measured in each time period t. The number of person-hours in each time period is the instantaneous count of people C_t multiplied by the number of hours in the time period L_t. The count C_t is conducted at some point during period t, either at the midpoint of period t or at a randomly selected time. A simple approach when many access points are involved is to move from one site to the next counting people and recording the time of day. Multiple counts at each access point (often taken over a period of several days) can be sorted into predetermined time periods, such as seven two-hour periods from 6 am to 8 pm. C_t for each period t can be estimated as the average of any number of counts falling in period t. Alternatively, the counts may be fit to a regression with dependent variable C_t and explanatory variables consisting of dummy variables representing different time periods, different sites, weekend/weekday, etc.

To estimate trips from total person-hours, one needs to know the average number of hours per trip. This can be estimated using on-site interviews. The most accurate approach is to intercept people as they exit a site and ask the number of hours they spent at the site that day. The average of individual responses gives the average length of a trip. Another approach is to interview people while they are using a site, going from person to person to obtain as many interviews as possible before proceeding to the next site. This roving technique again requires less effort, since an interviewer does not need to remain at a single access point waiting for people to exit. However, responses may not be as accurate because determining total trip length requires asking respondents to estimate their anticipated time of departure. Time spent at the site so far plus anticipated time to departure gives total trip length.

If the roving intercept method is used, the formula for average trip length is the harmonic mean of reported trip length rather than the arithmetic mean. The arithmetic mean cannot be used because the probability of selection is not equal across respondents. Specifically, the probability of intercepting a given respondent increases in proportion to the time the respondent spends at the site. To compensate for oversampling of longer trips, each observation must be weighted by the inverse of trip length. Average trip length $\bar{h}$ is calculated as

$$\bar{h} = \frac{\sum_n \frac{1}{h_n} h_n}{\sum_n \frac{1}{h_n}} \,. \qquad (3.7)$$

Expression (3.7) is a weighted average of trip length h_n for N respondents, where the weights are $1/h_n$. When calculating a weighted average, the weighted sum in the numerator must be normalized using the sum of the weights in the denominator (this is equivalent to dividing each weight by the sum of all weights, creating normalized weights that sum to one). Our formula differs from the formula given in Deacon and Kolstad (2000) because we have implicitly assumed that all interviews are conducted using a constant sampling rate. If different sampling rates are used, such as a complete enumeration in lightly visited locations and the selection of every third person in crowded locations, then the weights used in (3.7) would need to be adjusted to account for this additional effect on selection probabilities for observations of h_n. Expression (3.7) is sometime rearranged algebraically to resemble the standard formula for the harmonic mean of h_n:

$$\bar{h} = \frac{1}{\frac{\sum_n 1/h_n}{N}} . \tag{3.8}$$

Finally, the total number of trips T per day is calculated by dividing hours per day H by the average trip length $\bar{h}$. Helicopter overflights may be used to count people on the shoreline instead of on-site surveys, as in Byrd *et al*. (2001). Related methods for estimating the number and characteristics of recreation trips are presented in Malvestuto (1983), Pollock *et al.* (1994), Wallmo and Chapman (2003), and documents associated with the Tampa Bay spill (EERG, 1998).

3.3.6 Evidence of recreation impacts from resource contamination

Practitioners assessing impacts to recreational use often rely on evidence from previous studies. The scope of the available evidence is substantial, but many practitioners may be unaware of relevant studies contained in government documents or research reports. Relevant studies may be difficult to locate even if they are published in economic journals, because sometimes the focus involves methodological advances rather than the applied results of a valuation exercise. Below we provide a guide to past research on two important human-use topics in natural resource damage assessment: impacts of long-term contamination on recreational fishing, and recreational impacts from oil spills.

3.3.6.1 Contamination-related losses to recreational fishing

We have identified 20 studies that evaluate the impact of contamination on recreational fishing. Estimates of lost value differ considerably across studies due to variation in resource characteristics, affected populations, and levels of contamination. Differences are also due to a range of analytical assumptions, such as alternative model specifications or alternative estimates of the cost of driving. Researchers often report values in different forms, such loss per angler per year, loss per trip to any site in throughout the study region, or loss per trip to a particular site.

The most conceptually valid value estimate for comparison across models is an angler's willingness to pay for the absence of contamination conditional on taking a trip to a particular site. In the random-utility formulation, this is the ratio of two parameters appearing in the conditional indirect utility function describing choice alternatives. Specifically, this particular "per-trip" loss is calculated as the parameter representing lost utility due to the presence of contamination divided by the marginal utility of money. This per-trip loss was discussed previously in Section 3.3.3.2. The advantage of this measure of loss is that it is conceptually equivalent to a fee for site access, with the level of the fee set just high enough to reproduce the behavioral impacts of contamination. Other measures of loss are sensitive to the number of people living in proximity to a contaminated site, or the availability of nearby alternative fishing sites. Estimating a per-trip value that is analogous to an access fee nets out these other influences and directly captures the intensity of people's dislike for the presence of contamination.

Based on this consistent measure of value, the loss from contamination may vary from approximately $2.00 per trip to as much as $30 per trip (e.g., Montgomery and Needleman, 1997; Breffle *et al*., 1999; Parsons *et al*., 1999). MacNair and Desvousges (2007) found that much of the variation in literature estimates depends on alternative analytical assumptions. They developed a formal comparison of per-trip values using a selection of six literature studies and they adjusted estimation procedures in each study to reflect a single set of consistent assumptions. Based on their choice of assumptions they found that values ranged from -$1.59 per trip (indicating a positive value for contamination) to $9.18 per trip. The average value across the six studies was $5.17. A synthesis of results across a broader selection of studies is a worthwhile topic for further study.

The following list of references may assist researchers and practitioners in examining the impacts of contamination on recreational fishing. Each study contains at least one estimate of the value of fishing losses attributable to contamination, whether associated with the presence of a fish consumption advisory, an "area of concern" designation, or other indications of the presence of toxic contaminants. The references are Lavaca Bay Trustee Council (1998), Breffle *et al.* (1999), and Desvousges *et al.* (2000), which are publicly available reports; Phaneuf (1997), which is a Ph.D. dissertation; and 16 published articles including Jakus *et al.* (1997), Johnson and Desvousges (1997), MacDonald and Boyle (1997), Montgomery and Needelman (1997), Chen and Cosslett (1998), Jakus *et al.* (1998), Krieger and Hoehn (1998), Parsons and Hauber (1998), Phaneuf *et al.* (1998), Herriges *et al.* (1998), Krieger and Hoehn (1999), Parsons *et al.* (1999), Shaw and Shonkwiler (2000), Jakus and Shaw (2003), Morey and Breffle (2006), and MacNair and Desvousges (2007).

3.3.6.2 Recreation impacts from oil spills

Evidence of recreation impacts from oil spills consists primarily of studies measuring post-spill declines in the number of recreation trips. Evidence from these studies on the severity, geographic extent, and temporal duration of impacts from past oil spills is discussed below. In contrast to the numerous economic models evaluating the effects of long-term contamination on fishing, only two studies have used recreation-demand modeling to evaluate oil-spill impacts. For comparison to results cited in the previous section, it is worth noting that Hausman *et al.* (1995) obtained a per-trip loss of $3.27 for recreational fishing following the Exxon Valdez spill in Alaska. Their model evaluated impacts throughout the remainder of the year following the

March 1989 spill and addressed an area slightly broader that the geographic extent of direct physical oiling. English *et al.* (2004) estimated a per-trip loss of $9.70 for oil spill impacts to recreational shrimping following a spill in Charleston, South Carolina. The shrimping impacts involved several access points to the Charleston Harbor estuary with varying degrees of oiling. Because site-specific information about impacts on visitation is usually available, benefit-transfer studies evaluating spills typically do not rely on these per-trip values as a basis for recreation assessments.

The percentage decline in recreation trips is a useful measure of impacts following a spill. In most benefit transfer exercises, the measured percentage decline in trips is directly proportional to the assessed lost value. The decline in trips includes trips substituted to other locations and trips forgone altogether. In the American Trader case, government experts measured an 85% decline in beach trips during a five-week period immediately following the spill. These losses involved beaches along a 14-mile stretch of shoreline in the Los Angeles area that were closed for some or all of the five-week impact period (Chapman and Hanemann, 2001). During the subsequent two-and-half weeks, a 30-percent loss was in trips was estimated, and losses were believed to continue after that time but were not included in the assessment. English *et al.* (2004) measured a 32% decline in shrimping trips to Charleston Harbor over a five-week period when a spill occurred in the middle of the recreational shrimping season. The Ft. Lauderdale mystery spill in Florida caused a 15% decline in beach trips for one week (NOAA and FDEP, 2002); a spill in Chesapeake Bay caused a 10-percent decline in shoreline use including fishing, swimming, and picnicking over a six-month period (Byrd *et al*., 2001); and a spill in Puerto Rico led to a 30-percent decline in trips over a 60-day period (Tetra Tech, 2006). In most cases these figures reflect an average of losses that are initially more severe and then decline as the spill area is cleaned up and recovers.

The assessment for the Athos spill on the Delaware River evaluated impacts to shoreline recreation, waterfowl hunting, and boating. The loss of shoreline trips was estimated at 26% for the April to June period following the release of oil during the previous November. Impacts were divided into a high-impact area immediately adjacent to the spill, with a decline in trips of 31%; an area of moderate effects, with a 19% decline in trips; and an area of low impact with an 11% decline in trips. The overall decline in shoreline trips for the period June to October was 13%. Waterfowl hunting trips declined by 21% during December and January, the remaining two months of the hunting season following the spill. There was an 8% decline in boating activity throughout the summer season. For an assessment of losses to other types of recreation, see Clark *et al.* (1998), which estimated a decline in activity on walking trails following the Julie N oil spill, as well as a decline in ferry and tour boat trips, charter boat fishing trips, and whale watching.

In some cases modest oiling is observed on beaches but no impact to shoreline recreation is detected. In the North Cape spill, oil was observed on several beaches along Rhode Island Sound as late as April 1996 but by the summer beach season no effects on recreational activity could be detected (NOAA *et al.*, 1999). Light oiling was observed on a popular beach in Charleston, South Carolina near the end of the beach season in October, 2002, but a comparison to control sites suggested that recreational activity was not affected.

Impacts to recreation typically persist beyond the period when oil is apparent on the shoreline. In the American Trader case, cleanup was completed and all beach closures were lifted within five weeks after the spill, but a significant decline in attendance was still observed during the subsequent two-week period (Chapman and Hanemann, 2001). Post-cleanup impacts were thought to continue beyond two weeks, but these potential additional losses were not quantified. In many cases closures or advisories are minimal, occurring in only a few locations and lasting only a few days, while impacts to shoreline recreation are widespread and significant. Byrd *et al*. (2001) found that impacts to shoreline recreation in a tributary of Chesapeake Bay lasted up to six months and extended throughout a 15-mile stretch of river, even though shoreline closures applied only to beaches immediately adjacent to the cleanup-staging area and lasted less than one month. Intercept surveys conducted following the Athos/Delaware River oil spill indicated that oil continued to impact fishing on the Delaware River throughout the summer of 2005 following a spill that occurred nearly a year before in November 2004. The temporal extent of impacts from the Athos spill may have been due in part to the reappearance of oil that had sunk following the initial release and then resurfaced as the river warmed during the summer months.

It is common to include in the assessment area locations not exposed to oil but close to areas where oiling occurred because recreators might reasonably believe these areas to be oiled (Hausman *et al.,* 1995). Evidence from the Athos/Delaware River spill suggests that the geographic extent of impacts to recreation is closely associated with the physical presence of oil, but that the potential for oiling does appear to deter recreational activity in some areas that are clean. The on-site survey of shoreline fishing conducted for the Athos spill included locations just beyond the physical extent of oiling both upstream and downstream of the site of the spill near Philadelphia on the Delaware River. The river is tidally influenced and currents transported oil in both directions, but the oil was carried much further downstream than upstream. The result was that areas just past the upstream extent of oiling were still quite close the site of the spill, and surveys indicated that fishing in these areas was significantly affected. By contrast, areas just past the downstream extent of oiling were quite distant from the site of the spill, and surveys indicated that fishing in these areas was not impacted. In other assessments, researchers have found that members of the public do not distinguish between oiled areas and clean areas within readily defined geographic boundaries such as an embayment.

3.3.8 Non-monetary equivalence for recreational use

The scaling techniques for recreational losses described thus far rely on the explicit valuation of the use of resources for outdoor recreation activities. As an alternative, it is sometimes possible to identify metrics related to recreational value that can be measured more easily. Such an approach would be analogous to the service-to-service approach used in many HEA/REA models described in Section 2.0. When a metric for recreational value can be applied to both injury and restoration, the equivalence of losses and gains can be established.

One approach of this type compares trips lost due to resource injury with trips gained from restoration actions. In this case trips would be the scaling metric, and discounted annual trips would sum to zero across past and future periods. This type of application was first proposed in Unsworth and Bishop (1994) as a possible extension of HEA/REA methods, and has been applied in some NRD cases.

An innovative example of service-to-service scaling for recreation was developed for the Montrose NRD case in Southern California. Industrial releases of dioxins and other chemicals led to chronic contamination of ocean sediments along the Los Angeles coastline. Many species of fish popular with local anglers were deemed unsafe for consumption because they accumulate high levels of chemical toxins when feeding in ocean sediments. One method for restoring lost resource services involved the creation of artificial reefs. The reefs attract species of fish that do not feed in sediments and therefore remain clean enough for safe consumption by anglers. Using "clean fish meals" as a metric, the loss from contamination could be directly compared to gains from reef creation without relying on monetary values.

3.4 Accounting for cultural values in restoration scaling

In some cases cultural heritage plays an important role in defining the importance of natural resources. The issue of cultural values in natural resource damage assessment arises frequently in the context of claims by Native American tribes. While often settled outside the framework of restoration scaling (e.g., the *Exxon Valdez* oil spill and the 2003 oil spill in Buzzards Bay, Massachusetts) tribal claims have sometimes relied on nonmarket techniques, such as benefit transfer (Kuroshima Trustee Council, 2002). Interesting developments have also appeared in the nonmarket valuation literature, particularly involving indigenous communities in Canada and Australia (Adamowicz *et al*., 2004; McDaniels and Trousdale, 2005).

In what follows we describe some of the difficulties associated with conventional nonmarket valuation techniques when applied to the assessment of resource losses in indigenous communities. This is followed by a brief review the literature covering assessment-related issues, such as evaluation of contaminant exposure in tribal communities, the role of tribal governments in damage assessment, the collection of information on use of natural resources by indigenous people, and evidence on divergent preferences between indigenous and non-indigenous communities. Finally, two studies are described in detail to illustrate potential approaches to assessing tribal losses. The first example combines conventional revealed-preference and stated-preference methods with particular techniques that adapt these methods to a tribal setting. The second example illustrates a more complete departure from conventional techniques, presenting a set of choice scenarios that are evaluated orally and evolve in a group dialogue.

3.4.1 Difficulties with conventional nonmarket valuation

Standard nonmarket valuation techniques are based on willingness to pay for a resource improvement. For example, individuals are asked to estimate the most they would pay in higher taxes to ensure cleanup of a contaminated resource. This approach to determining compensation may present problems in the context of indigenous communities. One important issue is that willingness-to-pay questions imply indigenous people do not have a right to the resource in its pre-injury condition. The tendency of indigenous people to strongly reject this notion presents difficulties even for hypothetical willingness-to-pay scenarios (Trousdale, 1998; Adamowicz *et al*., 2004). A related issue is that monetary transactions may be of limited importance to some tribal communities, making the interpretation of willingness-to-pay scenarios difficult

(Adamowicz *et al*., 2004) and altering the way losses are perceived (Kirsch, 2001). Communal property and a consensus approach to decision-making may present problems for an individual-based willingness-to-pay study (Adamowicz *et al*., 1998b).

Another fundamental limitation is that willingness-to-pay methods are an inadequate approach to valuation when the resource being valued constitutes a large portion of a community's total endowment. For example, preventing damages to a resource could be more important to an individual than forgoing her total monetary income. In this case willingness-to-pay measures are of little help in analyzing value tradeoffs and determining appropriate compensation. Even if willingness to pay is less than total income, it is unlikely to be an adequate measure of compensation as long as it represents a significant share of total income. The reason is that giving up a large share of one's income as payment represents a more significant change in wellbeing than receiving the same amount of money as compensation. This and related issues are discussed further in Snyder *et al.* (2003).

Economic methods based on "willingness to accept" offer a potential solution to the problems associated with willingness-to-pay techniques. Willingness-to-accept methods directly elicit the level of payment required to induce an individual to voluntarily accept a loss. If correctly measured, this amount is necessarily equivalent to the appropriate compensation for resource losses. Natural resource trustees in the United States typically avoid willingness-to-accept methods due to concerns they may lead to damage estimates that are too high. Indeed, CERCLA regulations appear to specifically endorse willingness to pay as the appropriate measure of damages (DOI, 1986), as did the NOAA panel on contingent valuation (Arrow *et al*., 1993). Others have argued in favor of a willingness-to-accept approach (*e.g*., Bromley, 1995). The issue may be of particular importance in the context of tribal losses due to difficulties with the willingness-to-pay approach. A study by McDaniels and Trousdale (2005), which is described in detail below, illustrates the potential value of a willingness-to-accept approach when tied to issues familiar to indigenous communities.

3.4.2 Assessment in indigenous communities

In this section we provide a brief overview of literature which is not referenced elsewhere but which may inform the investigation of resource losses in indigenous communities. A study by Murray *et al.* (2005) examined the preferences of aboriginal communities in northern Saskatchewan. The study considered a set of "instrumental" values, referring to modes of behavior such as ambitious and clean, and a set of "terminal" values, referring to things that people seek in life such as equality, health and excitement. For members of the aboriginal communities studied, "honest" and "family security" were the top-ranked values while "logical" and "national security" were ranked at the bottom. The researchers also studied a local non-aboriginal community and found that differences between aboriginal and non-aboriginal groups were significant, but no greater than differences among aboriginal groups. The researchers also found differences between a representative sample of tribal members and the opinions of tribal leadership, derived in part from a lack of representation of women in tribal government.

Other studies also examined differences in preference between different tribes and between tribal and non-tribal communities. Haener *et al.* (2001) identified potential systematic differences in hunting preference between Metis and First Nations communities in Canada. Metis communities, which are of mixed aboriginal and European descent, tended to be more sensitive to travel cost and wildlife populations when choosing a hunting destination. First Nation communities appeared to be more interested in the hunting experience, and had a preference for water access to a site rather than road access. Loewen and Kulshreshtha (1995) found that aboriginal residents of Saskatchewan held higher values for wilderness protection compared to nonaboriginal residents. Rolfe and Windle (2003) and Adamowicz *et al.* (2004) both found that stated-choice tradeoffs could be used to learn about preferences in an aboriginal context.

DuBey and Grijalva (1994) analyzed the role of tribal governments in CERCLA damage assessment rules and proposed changes to address some limitations in tribal authority. Harper *et al.* (2002) conducted an investigation of human exposure to hazardous substances associated with subsistence activities in a tribal community living near a uranium-mine Superfund site. In addition to describing a wide range of subsistence activities and quantifying potential exposure, the authors emphasized the need for investigators to understand the tribal community and its history in order to successfully implement information-collection methods. Usher and Wenzel (1987) examined methods used to collect information on aboriginal harvest in Canada.

3.4.3 A study of indigenous resource use

Researchers in Canada assessed the value of hunting to indigenous communities in a study that examined timber harvest and non-timber values in forest management. They found that the preferences of indigenous hunters for resource attributes such as wildlife populations and landscape character could be modeled using conventional utility-based tradeoffs, but they also determined that data collection methods must be adapted for use in an aboriginal context. The results were reported in Adamowicz *et al.* (2004) and Haener *et al.* (2001).

The purpose of the forest-management study was to assist policy makers in accounting for the values of indigenous people when developing management plans for timber harvest. The study developed a site-choice model of moose hunting in northwestern Saskatchewan, where timber harvest takes place in areas used by aboriginal communities for hunting and other activities. The study relied on both revealed-preference data describing respondents' recent hunting activities, and stated-preference data based on hypothetical hunting destinations incorporating specific attributes described to respondents. Aboriginal groups in Canada include Inuit people of the Arctic, Metis people of mixed indigenous and European decent, and the remaining indigenous tribes of Canada, identified collectively as the First Nations. Inhabitants of the area addressed in the forest-management study included both Metis and First Nations people.

Standard data collection techniques were refined for use in an aboriginal context. Initial contacts were made with members of the community to obtain permission to conduct the study. Methods and objectives for the study were refined during the course of these initial contacts. For example, the researchers determined that focusing the study on moose hunting would appropriately capture an important component of the value of forests to indigenous tribes. These initial discussions therefore fulfilled many of the objectives that might normally be addressed during a more formal focus-group process. They also served to engage the community in the development and progress of the study.

Based on initial contacts, the researchers determined that the main data collection effort would use in-person interviews in order to adequately explain stated-preference choice alternatives to respondents. The interviews were initially designed as straightforward question and answer sessions, but evolved to resemble less formal conversations based on personal experiences and "story telling". To the extent possible, text-based questionnaires were replaced with photographs that illustrated the attributes investigated in stated-choice questions.

The researchers identified trust and reciprocity as key elements of the data collection process. Trust was established by employing a community resident to arrange interviews, and also to be present during the interview sessions to help put respondents at ease and assist with any language difficulties. All interviews were conducted by one primary interviewer, who lived and participated in the communities for a year during the course of data collection. The fact that the interviewer became known in the communities and developed relationships with local people was considered to be a fundamental part of the research process.

Reciprocity involved several components. First, information collected during the study was presented to each of the communities when the research was completed. The information included a map that summarized knowledge provided by respondents about special sites throughout the area, including moose calving areas, wilderness cabins, indigenous burial sites, and other locations of historic or cultural importance. Second, a gift certificate at a local hunting store was offered to all respondents. Third, in interviews with First Nations elders, an offering of tobacco was made at the start of the interview as a greeting and as a sign of respect.

A noteworthy aspect of the data collection and modeling was the use of respondent perceptions to estimate moose abundance. During the in-person interviews hunters identified, on a map, locations with high moose populations. Locations were defined based on forest-management operating areas delineated by resource managers. Explanatory variables in the model included landscape features within each operating area, such as density of rivers, disturbance by fire, and recent timber harvesting. Data on landscape features were drawn from the interviews as well as from maps of the area. A logit model generated an index of moose abundance based on the probability that landscape features at a given location would correspond to high moose populations as identified by hunters.

The revealed-preference model was estimated based on characteristics of the designated hunting areas and trips to each area as reported by hunters. The stated-preference model was estimated using responses to attribute-based choice experiments that offered respondents the selection between two hypothetical hunting destinations. A combined model was also estimated. While the revealed preference model indicated that hunters viewed recent timber harvests as a positive attribute, this was likely due to the positive effects of timber harvest on moose abundance and hunting access. The combined model indicated that recent timber harvests had a negative amenity value, while still indicating that hunters may prefer newly harvested areas through their effect on moose abundance. Separating these effects allowed the authors of the study to appropriately investigate the value of projects intended to benefit hunters in indigenous communities. For example, habitat enhancements may improve moose abundance without the negative amenity effects associated with timber harvesting.

One limitation of the model noted by the authors involved the difficulty of generating monetary values. While resource attributes identified in the model could be used to evaluate resource tradeoffs, the monetary evaluation of resource changes depended on converting the travel-distance variable into a monetary equivalent. The study used conventional travel-cost assumptions, namely, per-mile vehicle costs plus one-third of hourly income. One difficulty of this approach involved obtaining data on personal income. The researchers were told that it would be culturally inappropriate to collect income data from survey respondents. Imputing wage rates from employment data was not possible due to the inconsistent work patterns of many respondents. Instead the researchers used average income figures for the region obtained from the Aboriginal Census conducted by the national government. The researchers suggested that that the use of travel cost models for monetary evaluation would depend on additional investigation regarding the appropriate value of the travel cost variable.

3.4.4 A study of losses to tradition, community, and the environment

McDaniels and Trousdale (2005) describe an innovative technique for developing non-market values and compensation measures for a wide range of resource-related losses to indigenous people. The study examines impacts from petroleum development on aboriginal lands in Alberta, Canada. The central elements of the study are the following: (1) the use of oral histories obtained from tribal elders to help reconstruct baseline conditions; (2) the implementation of workshops in which community representatives reach agreement on cultural and environmental priorities; (3) the development of consensus-based relative weights to define value tradeoffs; and (4) the inclusion among the tradeoffs of financial gain from economic development, allowing for monetization of cultural and environmental losses. While some of these elements represent a departure from conventional practices and rigorous statistical methods, Canadian courts have upheld some aspects of the approach for litigation in an aboriginal context.

The study concerns compensation to Metis communities, settled starting in the early 1800s by traders of mixed indigenous and European descent. The *Metis Settlements Act* of 1990 established a legal framework for compensation to Metis communities in connection with petroleum development activities on Metis lands. In many cases petroleum development occurred with limited or no consultation with Metis people and was conducted in ways that were in conflict with local cultural and environmental values. In the case study examined by McDaniels and Trousdale, some funds from petroleum development were paid to Metis communities, but whether the funds were sufficient to compensate for losses was the subject of the investigation.

The first step in the study was to characterize the impact of petroleum development on Metis settlements. The tribes did not maintain written records and the most comprehensive source of information about historical conditions was oral histories related by tribal elders. The 1997 Supreme Court of Canada decision *Delgamuukw v. British Columbia* upheld the use of oral histories as evidence of historical facts. The information collected from tribal elders included geographic data on the extent of environmental and cultural impacts, which were compiled on maps of the region. It also included the identification of values and social conditions affected by development, often related in the form of stories. All information was validated in subsequent interviews and workshops.

The tribal elders identified the effects of development in four categories: traditional values, such as knowledge, skills, and spiritual sites; environmental or "bush" values, involving plants, wildlife, and respect for the land; community values, including health, safety, and community cohesion; and economic values, which referred to the income communities had received from petroleum development. Specific environmental impacts included noise, light, chemical spills, and landscape alterations causing poor drainage and flooding. Losses to traditional values included the degradation of traditional and spiritual sites, and the construction of roads and forest cut-lines leading to increased intrusions and poaching by outsiders. Community impacts included the uncertainty and anxiety associated with potential health impacts from environmental contamination. Economic benefits were equivalent to $325,000 in annual payments to community members.

The second step involved workshops conducted with community representatives. Workshop participants were chosen based on their knowledge of settlement affairs and their knowledge of issues related to petroleum development. They were also informed that the workshop objective was to determine appropriate compensation for petroleum development, and were selected based on their understanding of this objective. Workshops were conducted with seven to eight people and participants were selected to be representative of the settlements in terms of age, family status and gender.

Workshop participants were presented with information describing the changes due to petroleum development based on the interviews with tribal elders. The changes were described in terms of specific attributes that had been changed by petroleum development, relative to baseline conditions without petroleum development. There were four attribute groups that captured, respectively, the four types of changes: traditional values, bush values,community values, and economic values. The information was presented orally with the aid of pictures, maps, and quotes from Metis elders.

The third step required the development of tradeoffs among the four attribute groups. First, workshop participants were asked to identify the category of attributes that they would most like to change from actual conditions (with petroleum development) to baseline conditions (without development). The same question was applied to the remaining three categories, and so on. The identified hierarchy placed traditional values first, then bush values, community values, and economic values. Workshop participants were then asked to rate the relative importance of the four attribute categories. A series of marks was placed on a wall representing a scale from zero to 100. Zero represented something of no importance to workshop members, and 100 represented the importance of traditional values, the highest-ranked category of attributes. The three remaining categories were positioned on the scale to appropriately reflect their relative importance. McDaniels and Trousdale (2005) described this as an iterative process involving discussion of alternative positions and successive revisions, until consensus was reached. Relative to the value of 100 for changes to traditional values, the impacts to bush values received a rating of 85, impacts to community values received a rating of 60, and economic values received a rating of 30.

The researchers assumed an additive value function in which the ratings reflect weights applied to each of four binary outcomes, namely, the four attribute categories with and without petroleum development. Given this assumption, a value index representing negative impacts from petroleum development would be quantified at 245 (100 + 85 + 60 = 245). A value index of gains from petroleum development would be quantified at 30. The monetary value of negative impacts was then calculated by multiplying annual economic benefits of $325,000 by the ratio 245/30. The resulting total value of negative impacts from petroleum development was $2.6 million per year. Compensation was calculated by netting out the benefits already received, leaving $2.3 million per year.

For reasons of confidentiality, the figures reported in McDaniels and Trousdale (2005) do not represent findings from any specific study. However, the authors indicated that the results described were representative of several cases. It is worth noting that the sum of $2.3 million per year does not reflect individual willingness to pay summed over affected members of the community. Rather, workshop participants arrived at a collective agreement about impacts to group values and to resources held in common. A calculation of compensation for resource and cultural losses is implicit in the rankings developed during the group consensus process.

3.5 Impacts to navigation

The use of waterways for navigation provides significant economic value. In the United States alone, commercial waterborne navigation generates $7 billion in savings annually relative to other modes of transportation (USACE, 2000). Impacts to public waterways from pollution can disrupt navigation in two ways. Short-term events such as an oil or chemical spill can lead to the closure of navigation routes during cleanup operations. Long-term contamination in river or ocean sediments can preclude maintenance of navigable waterways at desired depths, because the presence of contamination increases the cost of dredging and disposal of contaminated sediments. When channel dredging continues despite contamination in sediments, the added cost of dredging and disposal also represents a quantifiable public loss.

Below we describe methods to address public losses that result from impacts to water resources used for navigation. We begin with an overview of consumer surplus and economic rent, which form the conceptual basis for losses associated with the disruption of navigable channels. We then discuss losses associated with commercial transportation, to our knowledge the only category of navigation-related damages that has been pursued in an NRD claim. We also discuss damages associated with recreational boating, another activity impacted by disruptions to navigation channels. Finally, we address damages associated with the management of dredged materials, specifically, the added cost of dredging and disposal of contaminated sediments. While the application of dredged material to beneficial uses such as marsh restoration is an important aspect of navigational dredging (e.g., MDNR, 2001), the loss of beneficial use has not been addressed in an NRD claim and is not discussed here.

All of the methods described below rely on the value-to-cost scaling approach, in which monetary losses are quantified and applied to appropriate restoration projects. To our knowledge recovered funds in past NRD cases have not specifically been applied to navigation-related projects, although such projects have been considered by NRD practitioners. For example, restoration could involve improving navigation channels using channel markers, implementing a system for gauging water levels and providing real-time information on channel depth, or dredging alternative navigation channels. Restoration could also include improvement of port facilities or support for rapid deployment of response actions following oil spills and severe weather events. Restoration involving resource enhancements could include wetland creation to slow the rate of sedimentation in rivers and harbors, reducing the required frequency of navigational dredging.

3.5.1 Consumer surplus and economic rent

Methods described up to this point, including metric-based HEA/REA models, stated-preference techniques, and travel-cost analysis, have been based on the measurement of consumer surplus. The value coefficients V_l and V_g introduced in equation (1.1) represent the public's willingness to pay for alternative levels of ecological and human-use services, which is equivalent to consumer surplus. The total value generated by natural resources also includes a second component known as economic rent. While consumer surplus expresses individual preference in monetary terms, economic rent shows up as actual monetary payments to participants in market transactions.

In Section 3.2.2 we described consumer surplus as the value of a good or service net of its cost. For a consumer, value is measured as maximum willingness to pay and cost is the price one actually pays. Economic rent is the analogous measure of net value for suppliers of an input to economic production. In the case of a supplier, the measures of cost and value are reversed. The price a supplier can obtain for an input represents value. The minimum payment that would induce a supplier to provide the input represents cost. The amount a supplier actually receives less the minimum amount she would be willing to receive determines net value, or economic rent.

As a consequence of scarcity in the supply of natural resources, commercial operations that benefit from the use of natural resources may generate economic rent. For example, the owner of an offloading facility for maritime vessels may be willing to provide access to the facility at a minimum price that earns a normal return covering operating costs and capital expenditures. However, coastal areas suitable for such a facility are scarce and limitations in the availability of access for maritime vessels might cause the value of access to increase substantially above this minimum price. If the port operator charges full value for use of the facility, she receives surplus value in the form of economic rent. Ports are often owned and operated by government entities, and the fees they charge may constitute less than full value. In this case economic rent accrues to vessel operators with access to the port, because they can take advantage of waterborne transport to move goods cheaply without passing the value on to the operator of the port or the owner of the resource.

Natural resource impacts to a waterway may increase the cost of vessel transport. For example, response actions following an oil spill sometimes force vessels to wait at sea or divert to other ports, and failure to maintain channel depth due to contaminated sediments can force shippers to use shallower-draft vessels at higher cost. The increase in cost leads to a reduction in the value of resource services that shows up in one of three places. First, consumer surplus could be reduced. This would be the case if the increase in cost is passed on to consumers in the form of higher prices for goods traveling through the port. Second, economic rent could be reduced. This would be the case if vessel operators pay the cost increase but do not raise prices, and if they are earning economic rent on their use of the port under baseline conditions. Economic rent could also be reduced in the form of a decline in user fees paid to the port facility. Third, the vessel operator may not be able to pass the cost increase on to consumers and may not be earning economic rent under baseline conditions. In this case the loss of resource services shows up as a reduction in profits for vessel operators to below the normal level of return.

3.5.2 Losses to commercial transport

Public claims for losses to commercial transportation were pursued in several cases in the United States, including the 1990 *Cibro Savannah* oil spill in the New York/New Jersey harbor, and the 1990 *Apex* oil spill in Galveston, Texas. Both claims involved the temporary closure of a navigation channel and used the same methodology to quantify losses, described in Julius (1992). The *Cibro Savannah* claim was upheld in United States district court (Cibro Savannah, 1996). Both claims predated the enactment of the Oil Pollution Act of 1990, which expanded the ability of private claimants to pursue damages for transportation disruptions.

Julius (1992) calculated the economic cost of a short-term closure of the Houston Ship Channel from July 28 to July 31, 1990. Using data on vessel traffic for the same period one year later, an estimate was developed of the number of vessels delayed by the closure and the additional operating costs incurred due to the delays. Based on the pattern of vessel arrivals and departures, it was determined that some vessels were trapped within the Port of Houston due to the closure, while others were forced to wait outside the navigation channel until the closure was lifted. The hourly cost of vessel delays was estimated based on hourly vessel operating costs and the conversion of capital and insurance costs to an hourly rate based on amortization of lump-sum amounts. The total number of hours that vessels were delayed was multiplied by the per-hour cost to calculate a total loss of $3.1 million due to the closure.

One difficulty with using the increased cost of transport as an estimate of natural resource damages is that compensable public losses consist of either consumer surplus or economic rent (DOI, 1986). Spill-related cost increases should not be part of a public claim to the extent that are borne by vessel operators in the form of below-normal economic returns. In addition, any portion of economic rent awarded to a private claimant must be deducted from the public claim (DOI, 1986; Jones *et al*., 1996). Difficulties in separating out the various components of loss were addressed in Julius (1992) using the assumption that all increased costs were passed on to consumers in the form of higher prices. The quantified damages thus represented a decline in consumer surplus. This assumption may be justified if the shipping industry is characterized by constant long-run average costs and if the periodic occurrence of navigational disruptions is fully anticipated, as argued by Julius (1992).

Damage quantification based on an increase in transport costs can also be applied to a long-term loss of navigation services. Long-term losses typically occur when contaminated sediments cannot be dredged and navigation channels are not maintained at optimal depth. The use of shallow draft vessels or the diversion of deep-draft vessels to alternative ports entails an increase in transport costs. While this may lead to below-normal profits for some vessel operators in the short run, long-run competitive forces are likely to cause firms earning below-normal returns to exit the market. This means that all increases in transport costs end up as reductions in economic rent or reductions in consumer surplus and are therefore subject to a public claim. In principle, this simplifies the quantification of losses in the case of long-term contamination. The primary limitation would be an overlapping private claim for lost economic rent, and if the value of such a claim is known it can be deducted from the assessed public losses.

To our knowledge no studies have examined long-term losses to commercial transport at NRD sites. However, numerous benefit-cost studies have been conducted by the Army Corps of Engineers to analyze channel-deepening projects (HDR Engineering, 2000; USACE, 2002, 2004). Estimating the benefits of a channel deepening project is the same analytical exercise as estimating the loss of navigational services when channel depth is not maintained due to contamination. HDR Engineering (2000) describes a model of the origin and destination of goods throughout the region served by the Port of Portland, Oregon. Transport models based on the origin and destination of goods can be used to estimate the loss of navigational services when a decline in channel depth forces shippers to switch to alternative ports or alternative railroad or trucking routes. USACE (2002, 2004) analyzed the use of vessel lightering (cargo offloading) on the Delaware River navigation channel. Vessel lightering and the use of shallower draft vessels allow shippers to enter navigation channels that are not maintained at depths suitable for deep-draft vessels. The additional cost of vessel lightering represents another potential approach to assessing commercial navigation losses.

3.5.3 Recreational losses

The dredging of harbors and channels benefits recreational boaters, and many navigational channels are dredged primarily for recreational boating. Short-term interruptions to recreational boating caused by oil spills have been evaluated in numerous cases using techniques described in Section 3.3, including travel-cost models, stated-preference methods, and benefit transfer (e.g., Breffle *et al*., 1999; Byrd *et al*., 2001; Athos 2007). When contaminated sediments prevent the dredging of recreational channels and cause long-term losses, the same techniques can be applied. For example, Whitehead *et al*. (2007) conducted a stated-preference survey evaluating the willingness to pay of recreational boaters for a program to dredge and maintain the Atlantic Intracoastal Waterway in North Carolina. The suitability of a recreational waterway for larger boats requiring a deep channel is a site characteristic that can be evaluated in a multiple-site travel cost model. If estimates are available of the decline in trips resulting from failure to maintain a recreational channel, then benefit transfer can be applied using a per-trip value for recreational boating obtained from the economics literature.

Losses to recreational boating may take the form of economic rent. For example, zoning and land-use restrictions may limit the number of marinas providing access for boaters, allowing marina owners to obtain above-normal economic returns. The issue is conceptually similar to the case of commercial transport facilities described above. However, the methods available for assessing losses to recreation differ from those applied to commercial transport. The most readily available methods for assessing losses to commercial transport involve an increase in cost resulting from resource impacts. Once the increase in cost is determined, the loss must be apportioned between consumer surplus, economic rent, and private losses. In the case of recreational use, the most reliable assessment methods directly address losses to consumer surplus, as described in Section 3.3. While in some cases it may be worth pursuing a public claim for lost economic rent not recouped by private claimants, the great majority of losses are likely to be addressed using standard recreation assessment techniques.

3.5.4 Added cost for disposal of dredged materials

If navigation channels are impacted by contaminated sediments but have nevertheless been maintained at depths consistent with baseline conditions, there would be no interim loss of navigation services. However, it is likely there would instead be losses due to increased dredging and disposal costs. The additional costs attributable to contamination can vary significantly depending on local handling and disposal options, and may vary from a few dollars per cubic yard to $20 per cubic yard, or more. The process for quantifying added costs includes identifying past dredging projects and commitments for future dredging, determining total quantities of dredged materials, evaluating the cost of likely disposal options under baseline and actual conditions, and multiplying dredged quantities by the appropriate unit added cost.

3.6 Impacts to commercial harvest

Resource impacts can lead to a decline in fish populations available for harvest by commercial fisherman. Impacts can also involve restrictions on the commercial harvest of fish that are contaminated or potentially contaminated. Losses in these circumstances are described in Jones *et al*. (1996). The situation is analogous to a disruption of navigation, described in the previous section, in that the nature and extent of public losses are determined by private markets. Economic damages from impacts to commercial harvest may include lost consumer surplus, lost economic rent, and a decline in private economic returns.

Lost consumer surplus would result from an increase in the market price of fish due to supply reductions. Consumers either pay the increase in price or purchase less fish, reducing the net value to the public of the commercial fish market. Lost economic rent would result from a decline in the harvest and sale of fish by commercial fisherman. The value of the commercial harvest to fisherman can include economic rent if fishing effort is constrained under baseline circumstances. As noted by Jones *et al*. (1996), a well managed fishery generates economic rent by preventing overexploitation that occurs in a competitive market. Restrictions on overexploitation of the fishery lead to scarcity in the form of limited access, and the result is a more productive resource than would prevail under open competition. In many cases rents are dissipated under baseline conditions through excessive participation in commercial fishing. In this case contaminant-related losses could lead to below-normal economic returns for commercial fisherman.

Unfortunately, there are no examples of the assessment of public losses for commercial harvest available in NRD cases. Commercial fisherman often bring private claims, which may capture economic rent as a component of lost profits. However, Hanemann and Strand (1993) argue that the actual level of economic rent generated in most commercial fisheries is a poor measure of damages. They argue that the appropriate measure of damages is the optimal level of economic rent, which is not realized in most fisheries due to poor management and would therefore not be assessed in a private claim. Quantifying optimal rents may be difficult in practice. Lost consumer surplus is also difficult to measure reliably because any change in market price is likely to be small. Changes in price might be measurable if significant impacts occurred to a specific type of fish representing a distinct segment of the market.

Impacts to commercial fisheries may also involve a consumer surplus loss from perceptions of degraded quality. Unlike fishing closures that lead to supply disruptions affecting the quantity or price of fish, public perceptions of a decline in quality lead to a downward shift in consumer demand. A stated-preference study could estimate damages based on the public's willingness to pay to prevent contamination-related impacts to the quality of commercially sold fish.

4.0 Discounting

There are two steps in establishing the equivalence of lost and restored resource services. The first step involves equivalence in the type and quality of resource services lost and restored. This has been the topic of previous sections. The second step involves equivalence across different time periods in which service losses and gains occur. There is inevitably a delay between the onset of injury and the completion of compensatory actions, and economic theory is clear that the delay matters when determining the adequacy of compensation. Equivalence of resource services across time is addressed through economic discounting.

It is worth noting at the outset that methods for discounting environmental benefits are difficult to define precisely and are the subject of considerable debate. Discounting reduces the measured importance of future benefits, and the selection of alternative discounting methods has dramatic consequences for policies affecting the distant future, such as climate-change mitigation. Sources of uncertainty include the role of intergenerational equity (Portney and Weyant, 1999), a lack of knowledge about the value of relevant variables such as interest rates and economic growth in the distant future (Weitzman, 2001; Newell and Pizer, 2003), and uncertainty about which variables are most relevant even today (Freeman, 2003; Moore *et al*., 2004). An important figure in the development of discounting theory once commented that the economic processes involved are too complicated and unfamiliar to feel comfortable with the choice of a discount rate without first examining the implications of alternative choices (Koopmans, 1965).

Despite the difficulties, there is little disagreement that economic discounting is essential to evaluating environmental costs and benefits that occur across different time periods, and a choice must be made about how to do it. A common practice has emerged in the field of damage assessment, and as a result this final step in determining appropriate resource compensation is relatively straight-forward.

4.1 The reason for discounting

The basic logic of discounting can be illustrated by comparing the value of an acre of marsh this year and next year. Suppose a person is offered the immediate restoration of one acre of marsh this year, and she places a value of $1.00 on this offer (this is her maximum willingness to pay to obtain the benefits of marsh restoration immediately). If conditions are essentially the same next year as this year, then when next year comes the same individual considering the same offer will again arrive at the same value of $1.00 for immediate march restoration. Now consider two alternative offers proposed in the same year. Specifically, there is a choice to be made this year between restoring one acre of marsh this year and restoring the same acre of marsh next year. What is the value of each alternative? We know that the value of restoring the marsh this year is $1.00. However, it would not make sense for the decision-maker to be willing to pay $1.00 now for the restoration of one acre of marsh next year. After all, she knows she will be willing to pay $1.00 for the restoration project next year, when the restoration actually occurs. If she declines to pay the dollar now, she can put the money in the bank and, at an interest rate of 3 percent, by next year she will have $1.03. Giving up $1.00 this year is equivalent to giving up $1.03 next year, and $1.03 exceeds next year's value for the marsh restoration project by three cents. This means that her willingness to pay this year for marsh restoration next year must be less than $1.00. More precisely, marsh restoration next year has a present value of $0.97 (or $1.00 divided by 1.03), because paying $0.97 now is equivalent to paying $1.00 next year.

Three assumptions in the above discussion should be addressed. The most important is the adoption of a 3-percent interest rate from the financial markets as the most appropriate discount rate for resource planning decisions. This will be addressed in detail below. The second assumption is that conditions are the same from one year to the next, leading to a constant value for resource improvements at any selected point in time. This may not be the case for a variety of reasons. One important reason is that the extent of natural habitats such as marsh may be declining over time, leading to an increase in their marginal (per-acre) value. This would suggest that willingness to pay for marsh restoration in the future may be higher than it is today (the assumption of a $1.00 constant value at any given point in time would have to be revised upward). As a matter of theory, this revision affects only the value to which the discount rate is applied and does not alter the discount rate itself (Fischer and Krutilla, 1975). As a practical matter, the effect of changes in willingness to pay is unlikely to be a significant factor for many injury and restoration calculations that occur over relatively short periods of time. In cases where restoration benefits are expected to extend many years beyond the period of injury, the potential impact of increasing values over time is likely to be offset by uncertainties in restoration success, which reduce the expected value of restored resource services over time (Hampton and Zafonte, 2004).

The third assumption is that environmental values can be expressed in monetary terms. While economists take this notion for granted, they also place limitations on the monetization of environmental values. Specifically, they ascribe monetary value only to non-essential environmental goods (Bockstael and McConnell, 1999; Bockstael *et al*., 2000). This means that marginal changes affecting the quality of human experience can be valued, but that large-scale losses, such as those potentially affecting the viability and sustenance of human life, cannot be valued. Recognizing this limitation may help allay concerns of some skeptics regarding the ability to monetize environmental values. To address any remaining skepticism, it should be sufficient to point out that all people cause damage to natural resources on a routine basis through their consumption of electricity, raw materials, land, and food, and it is therefore clear that we are willing to sacrifice some measure of environmental quality for the sake of personal comfort. Any suggestion that natural organisms or communities have rights that supersede our ability to put a price on their health or survival is contradicted by the recognition that there is evidently a price at which we are willing to ignore those rights in favor of our own self interest.

While the monetary evaluation of environmental benefits is critical to analyzing injury and restoration over different time periods, it is not necessary to make the monetary values explicit. We know from the above analysis that an acre of marsh lost this year can be replaced by 1.03 acres of marsh restored next year. We know this because the present-value loss is the $1.00 value of the marsh, and the present-value gain from next year's restoration is also $1.00 (calculated as next year's value of $1.00 multiplied by 1.03 acres divided by a discounting adjustment of 1.03). The marsh could be valued at $1.00, $5.00 or $100, and this adjustment in value would not affect the determination of how much marsh restoration is required. The detour through a money-based analysis simply allows us to take advantage of information in the financial markets showing how people compare benefits and losses across different time periods.

We have used the existence of positive interest rates to conclude that a given increment in future benefits must be worth less to people than the same increment in current benefits. To round out the story, it is worth knowing why this might be so. Economists attribute the preference for near-term benefits to two factors. The first is impatience, or "pure time preference". This concept takes as given that all else being equal, people prefer immediate benefits over delayed benefits. The second factor relates to growth in income and wealth. As people acquire greater wealth over time, the value of each increment in wealth declines. A detailed account of the theory of economic discounting is presented in Frederick *et al.* (2002) and Heal (2005). A practical and interesting article explaining discounting and the selection of an appropriate discount rate is Moore *et al.* (2004).

4.2 The appropriate discount rate

Controversies over the selection of an appropriate discount rate have been largely overcome in the field of damage assessment. The use of a 3-percent discount rate is the accepted practice, having been consistently adopted in hundreds of NRD settlements. Support for the use of a 3-percent rate for addressing environmental values can be found in damage-assessment guidance documents issued by NOAA (NOAA, 1997, 1999); an "issue statement" released by DOI on social policy analysis (DOI, 1995); and guidance from the Office of Management and Budget and Environmental Protection Agency on the development of economic analysis (OMB, 2003; EPA, 2008). The 3-percent rate has frequently been adopted when addressing topics outside the field of NRD but closely related to it, such as the evaluation of ecosystem services for land-use planning (NJDEP, 2007). When an academic survey asked more than 2,100 leading economists to estimate the appropriate discount rate for long-term environmental projects, the mean response was 4 percent, the mode was 2 percent, and the median was 3 percent (Weitzman, 2001).

Methods for selecting an appropriate discount rate involve a comparison between characteristics of the good being evaluated and characteristics of analogous investment instruments traded in financial markets. The key characteristics affecting interest rates are volatility in the flow of benefits (or risk), and the time period over which benefits accrue. Most researchers suggest that a risk-free interest rate should be used to evaluate public projects like habitat or resource restoration. The primary argument is that public entities undertake a large number of diversified projects and that the risk of lower-than-expected benefits from some projects are offset by higher-than-expected benefits from other projects (Arrow and Lind, 1970). This argument may not be well suited to restoration projects explicitly meant to compensate for lost resource services impacting specific communities. On the other hand, the use of financial-market risk as an analogue to uncertainties associated with environmental benefits may be limited. This suggests that project risks are more appropriately addressed as reductions in estimates of expected benefits, rather than as adjustments to the discount rate (Boardman *et al*., 2001).

The standard source of information on risk-free interest rates is the market for United States Treasury notes. However, interest rates for U.S. treasuries vary significantly depending on time to maturity. Long-term notes pay higher interest rates than short-term notes. The analogy to preferences for environmental services is again limited because differences in long-term and short-term interest rates are partly explained by a "liquidity premium" for long-term instruments, reflecting the additional risk associated with postponing repayment of the notes. This may have little relevance to the value of postponed enjoyment of environmental services. However, most analysts appear to accept that long-term interest rates provide the most accurate representation of tradeoffs people are willing to make between public benefits in different time periods (OMB, 2003; Moore *et al*., 2004).

Two final considerations help determine the most appropriate discount rate. One is the effect of taxes, the other the effect of inflation. Individuals who purchase U.S. treasuries do not receive the interest payments in full, but instead receive after-tax payments. The willingness of consumers to save at prevailing interest rates indicates a willingness to trade current value for future value at the after-tax rate. In a similar way, annual inflation reduces the value of future returns. The reduction in the effective return of financial instruments once taxation and inflation are accounted for reduces the magnitude of the appropriate discount rate (DOI, 1995; Freeman, 2003; Moore *et al*., 2004).

The historical average of the real (inflation-adjusted), after-tax interest rate on 10-year U.S. treasury notes is approximately 2 to 3 percent. This figure is reported in Freeman (2003), and is also obtained by reducing the pre-tax average rates reported in DOI (1995) and OMB (2003) by an average marginal tax rate of 30 percent (Moore *et al.,* 2004). Evidently the commonly accepted 3-percent rate may be slightly high according to the prevailing theoretical arguments. However, it may be justified in the case of resource compensation if uncertainties in the flow of restoration benefits are not fully addressed in restoration scaling metrics (NOAA, 1999; Hampton and Zafonte, 2004). Importantly, adherence to established guidelines resolves at least one source of uncertainty in the calculation of natural resource damages, and the practice of using a 3-percent discount rate is likely to continue for the foreseeable future.

4.3 Discounting methods

Discounting is typically done on an annual basis. However, environmental benefits do not occur at some specified point in time each year, but instead occur continuously. The most appropriate way to evaluate continuous benefits is to use mid-year discounting (OMB Circular A-94). Mid-year discounting closely approximates the precise discounting calculation, which is a continuous integral. Based on a 3-percent interest rate, mid-year discounting starting in year 1 and extending through year T employs the discount factors given below.

$$\text{Discount factors for years 1 through } T\text{:} \quad \left\{ \frac{1}{1.03^{0.5}}, \frac{1}{1.03^{1.5}}, \frac{1}{1.03^{2.5}}, \ldots, \frac{1}{1.03^{T-1.5}}, \frac{1}{1.03^{T-0.5}} \right\}.$$

Table 4.1 illustrates discounting calculations for hypothetical injury and restoration service flows extending over 100 years. The mid-year discount factor declines from $1/1.03^{0.5} = 0.985$ in year one to $1/1.03^{99.5} = 0.053$ in year 100. Losses from injury and gains from restoration are both expressed in percentage terms on a unit-area basis. Service flows follow linear trajectories. At the beginning of year 1, the loss per acre from injury is 100% of the full service level at the injury site. Losses decline to 50% by the second year and 25% by the third year, with full recovery and no losses in subsequent years. The gains from restoration are also expressed as a percent of full service at the injury site. Injury and restoration may take place in the same habitat type, or the calculation of restoration gains may involve the use of habitat conversions. For example, restoring services by 40% in a habitat whose full value is half that of the injury habitat results in a 20% service increase relative to the injury site. In the example below, restoration begins in year 2 (one year after the initial injury), with services increasing from 0 to 10% by the beginning of year 3 and increasing to 20% by the beginning of year 4. Restored services remain at 20% through year 100.

Mid-year discounting requires the conversion of service flows to mid-year values. For example, service losses from injury in the middle of year 1 are (100% + 50%)/2 = 75%. Mid-year service flows are multiplied by the respective mid-year discount factors to obtain mid-year discounted service flows. The 75% service loss in the first year of injury is multiplied by 0.985 to arrive at a 74% discounted loss. By year 100 the 20% service increase from restoration reduces to a 1% service gain in discounted, present-value terms. Discounted per-acre service flows are added up across years to calculate discounted service acre-years (DSAYs) for losses due to injury and gains from restoration. Discounted services for injury sum to 121%, shown in the final row as a total of 1.21 DSAYs of injury per acre. One acre of restoration generates 6.03 DSAYs of benefit.

Table 4.1. Discounted service flows

		Annual service losses from one acre of injury			Annual service gains from one acre of restoration			
Year	Mid-year discount factor	Beginning of year	Mid-year	Mid-year discounted	Beginning of year	Mid-year	Mid-year discounted	Gain/loss ratio
1	0.985	100%	75%	74%				
2	0.957	50%	38%	36%	0%	5%	5%	
3	0.929	25%	13%	12%	10%	15%	14%	
4	0.902	0%	0%	0%	20%	20%	18%	
⋮	⋮	⋮	⋮	⋮	⋮	⋮	⋮	
99	0.0544	0%	0%	0%	20%	20%	1%	
100	0.0528	0%	0%	0%	20%	20%	1%	
Total DSAYs				1.21			6.03	5.0

The final column in Table 4.1 shows a 5.0 ratio of gains to losses. This means that each acre of restoration generates benefits five times as great as the losses that accrue from each acre of injured habitat. This ratio can be applied to the area of injury to calculate restoration requirements. For example, a 10-acre injury site requires compensation of 10/5 = 2 acres of restoration, based on the injury and restoration parameters presented above.

5.0 The future of restoration scaling

Restoration scaling has a relatively short history of only about three decades. The field has been created by economists and ecologists in response to mandates of federal environmental laws in the U.S. Specifically, the Comprehensive Environmental Response Compensation and Liability Act (CERCLA), the Oil Pollution Act (OPA), and the Clean Water Act (CWA) each required responsible parties to compensate for injury through restoration or mitigation that replaced quantitatively the losses. NOAA has been the lead federal agency in overseeing development of this field of compensatory restoration, although other federal agencies, several states, and other nations in the EU are now playing growing roles. The development of the conceptual basis for the field has coincided with and facilitated the evolution of the fields of natural resource economics and of restoration ecology. Thus, in many ways practical demands from government have done much to create new academic directions.

A sufficiently large body of research and application has been produced over this three decades that we judged the time to be ripe for a broad overview of the field of restoration scaling. That was the motivation that brought us together to write this synthesis. Many academic and agency scientists, both economists and ecologists, are relatively unfamiliar with this new field. Our hope in synthesizing the body of work that has been produced and making it more readily available by assembling and making readily accessible through the NOAA Coastal Resources Research Center at the University of New Hampshire many unpublished documents describing applications to cases is to stimulate development of new approaches and techniques even better than the current methods.

We can anticipate some of the new directions in which restoration scaling may evolve (Peterson and Lipcius, 2003). Clearly, research is progressing at the interface of ecology and economics that not only identifies ecosystem services explicitly but also provides quantitative economic valuations for those services. Such developments can lead to more refined metrics of injury and can serve to improve the economic valuation of injury, useful in those cases where compensation is monetary rather than provided through natural resource restoration projects. As more opportunity to conduct follow-up evaluations of natural recovery and ecological development of newly restored habitats, the assumptions underlying temporal development of ecosystem functions are directly tested (as in Michel *et al.,* 2008). Follow-up testing will improve such assumptions in future scaling applications. In aquatic systems, especially those with moving water like tidally driven estuarine and riverine ecosystems, connectivity among habitats is great. This implies that the habitat value for any given habitat type is not fixed but varies with landscape setting. Such spatial proximity considerations would affect injury quantification as well as services arising from habitat restorations. Such considerations enter into the injury assessment and restoration scaling for the Hybelos Waterway (Commencement Bay Trustee Council, 2002), but similar approaches are likely to be developed and more routinely applied. As climate change progresses, and as sea level increases, impacts on estuarine habitat functioning are likely to be large, modifying greatly how restoration scaling is computed.

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