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# The role of ecosystem services in USA natural resource liability litigation

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A B S T R A C T

This paper examines how the United States has valued harm to public resources in natural resource liability laws and practice, an early legal application of the ecosystem-services conceptual framework. Our primary focus is on valuing harm to the difficult-to-value resources and ecological services that provide indirect or passive human uses, for which revealed preference valuation methods (based on observable behavior) are not applicable. We concentrate on the past 25 years of U.S. experience with the innovative, restoration-based framework established in regulations implementing the Oil Pollution Act of 1990. By reframing the damage claims as the cost of both "primary" restoration (to promote recovery of injured resources) and "compensatory" restoration (to account for interim losses pending recovery), the regulations deflected some of the controversy surrounding valuation methods.

The restoration-based compensation framework provides two basic approaches for calculating the scale of compensatory restoration projects. A *service-to-service* approach, which does not require valuation, applies to projects that provide resources and ecosystem services of the same type, quality, and comparable value as those harmed. A *valuation approach*, intended for a broader range of applications, relies on survey-based methods.

For injuries to ecological services, we found trustees have relied almost exclusively on habitat equivalency analysis (HEA), a service-to-service approach, adapting its use to applications where restoration projects make resource and/or ecosystem services substitutions. We explore how the trustees address the challenge of characterizing the equivalency between injury and restoration resources and ecosystem services through the choice of restoration projects and the choice of the ecosystem service metrics. Widely used in the U.S. and EU, the restoration-based measure of damages and the associated HEA methodology may be useful for other countries.



## 1. Introduction

Traditionally, legal claims for natural resource damages in common law systems required physical injuries to a private property interest and restricted recoveries to financial losses associated with market goods. Both requirements are particularly limiting for cases involving harm to natural resources in the public domain (Lee and Bridgen, 2014; Ward and Duffield, 1992).

Over the past 40 years, a growing number of countries have adopted natural resource liability statutes with innovative provisions that surmount both of these restrictions in order to address environmental harms from oil spills, toxic contamination, illegal development and other sources (Jones et al., 2015; Percival, 2010; UNEP, 2010). One critical innovation has been to establish legal standing for various organizations to file claims for damages to public natural resources. The government agencies that own, manage, or protect the resources are typically designated standing to file claims; in many countries, communities and civil society organizations are also granted standing. Another critical legal innovation has been to value natural resource damages beyond market-based losses and to expand the set of allowable methodologies to include non-market valuation methods.

Driven by the demand created by the enactment of numerous environmental statutes, the past 40 years has witnessed development of a framework and methodologies to value natural resources

*Abbreviations:* CERCLA, Comprehensive Environmental Response, Compensation and Liability Act; EU, European Union; HEA, habitat equivalency analysis; NOAA, National Oceanic and Atmospheric Administration; NRDA, natural resource damage assessment; PRP, potentially responsible parties; OPA, Oil Pollution Act; US EPA, US Environmental Protection Agency; US DOI, US Department of the Interior.

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as natural capital providing a range of market and non-market service flows (Daily, 1997; Freeman, 1993; Freeman et al., 2014; Gómez-Baggethun et al., 2010; Guerry et al., 2015; Kopp and Smith, 1993; Ward and Duffield, 1992). While natural resource valuation has been used mostly to inform ex ante benefit-cost valuation of proposed policies and projects, it has also been used for ex post valuation of natural resource injuries in order to claim damages for litigation (Kopp and Smith, 1993). As liability statutes for public resource injuries have been adopted worldwide, the language in these statutes and regulations at various points in time reflects the evolution of the natural resource valuation and the complementary ecosystem services and environmental economics literatures.[1] A major challenge in implementing the natural resource liability provisions, however, has been valuing damages to ecological services with indirect and/or passive human uses–including regulating services (e.g., floodwater storage and conveyance, and climate regulation) and habitat services (e.g., nursery services and gene pool protection)–for which valuation methods that rely upon observable behavior are not applicable (ibid.).

In this paper, we focus on the Oil Pollution Act of 1990 (OPA) and the subsequent 25-plus years implementing natural resource liability in the U.S. The 1990 act contains the most complete statement of public (and private) liability provisions, and the 1996 implementing regulations incorporated an innovative restoration-based framework for valuing damage claims that avoided the contentious use of stated preference methods for valuing indirect and/or passive human uses. The restoration-based framework has been widely adopted in the U.S. and EU.

OPA's full compensation approach for damages to public natural resources includes restoring or replacing injured or destroyed resources, compensating for interim losses pending recovery, and recovering the cost of the assessment (33 U.S.C. 2706(d)(1)). The natural resources damages provision complements additional OPA provisions that enable separate claims for private losses to real or personal property, profits and earning capacity, and subsistence use; and for public losses to revenues or increased costs (33 U.S.C. 2702(b)(2)(A)–(F)).

The issue of natural resource valuation for damage claims was very contentious when the National Oceanic and Atmospheric Administration (NOAA) was developing regulations to implement the natural resource damage assessment (NRDA) provisions of OPA (Portney, 1994). Valuation methods in general, and contingent valuation in particular, had also been controversial during the development of NRDA regulations for the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) (Kopp and Smith, 1993; Ward and Duffield, 1992). But the U.S. Congress enacted OPA after the 1989 Exxon Valdez oil spill, an environmental disaster that despoiled a pristine environment in Alaska. The *Exxon Valdez* natural resource liability litigation, in which contingent valuation was used to value the damage claim, alerted potentially responsible parties to the significance of the valuation issue (Jones, 2000).[2] The central elements of the controversy were (1) whether to include damages for lost ecological services that provide indirect and/or passive human uses and (2) whether to allow the use of stated preference methods (with a focus on contingent valuation at the time) to estimate such uses (ibid.).

NOAA reframed the interim loss component of the damage claim from one of monetary compensation (how much money does the public require to make them whole?) to one of resource compensation (how much compensatory restoration does the public require to make them whole?). The reframing of the measure for interim loss compensation is consistent with the statutory mandate that all recoveries for natural resource damages are to be spent on restoring injured resources and/or acquiring equivalent natural resources (33 U.S.C. § 2706(f)). The regulations formalized an ongoing shift in trustee practice away from monetary valuation of interim losses toward resource valuation. By recovering the costs of restoration as the damage claim rather than interim lost value—and thereby de-emphasizing the role of valuation methods—this framework is recognized by various stakeholders as a less controversial way to litigate damages to ecological services.

The restoration-based compensation framework provides two basic approaches for calculating the scale of compensatory restoration projects. A *service-to-service* approach—a simplified technique analogous to in-kind trading, which does not require valuation—was designed for compensatory restoration projects that provide resources and ecosystem services of the same type and quality, and comparable value, as those injured. Habitat equivalency analysis (HEA) is the predominant method for implementing the service-to-service approach. A *valuation* approach, intended for a broader range of applications, relies on survey-based stated preference methods to value the tradeoffs between environmental losses and prospective compensatory restoration projects.

In this paper, we look back at practices used by U.S. natural resource trustees in implementing the resource compensation measure for harm to ecological services over the 25 years since OPA was promulgated. In Section 2, we provide an overview of the key U.S. statutes with natural resource liability provisions, and then highlight the key OPA provisions pertaining to development of a damage claim. In Sections 3 and 4, we focus on how trustees have implemented the service-to-service and valuation approaches to scaling restoration-based compensation for lost ecological services, with particular attention to the difficulty of characterizing equivalency in ecosystem services at injury and compensatory restoration sites. We concentrate on HEA, the major service-to-service approach and the predominant method used for scaling lost ecological services. In Section 5, we discuss advantages and critiques of resource compensation in general and HEA in particular, as well as impediments to full-scale adoption of the new class of production function-based ecosystem service models. We also consider the use of restoration-based approaches in natural resource liability in other countries. An Appendix provides a case study of a natural resource damage claim for mining contamination impairing protected salmon habitat, including a status report on restoration outcomes.

## 2. U.S. natural resource liability statutes

### 2.1. Overview of U.S. statutes

In the U.S. common law system, the body of substantive environmental law includes well-articulated statutes and regulations for determining and measuring natural resource liability. These statutes and regulations have influenced legislation globally (Goldsmith et al., 2014; Percival, 2010). The U.S. statutes containing resource liability provisions typically cover risky activities or protected resources (Lee and Bridgen, 2014; Ward and Duffield, 1992). The Comprehensive Environmental Response, Compensation and Liability Act of 1986 (CERCLA, more commonly known as Superfund), the Federal Clean Water Act (CWA) Amendments of 1977 and the Oil Pollution Act of 1990 (OPA), all focus on oil

[1] In the U.S., the initial implementing regulations for CERCLA (1986, 1987) referred simply to "resources" (40 CFR § 300.3), whereas the implementing regulations for U.S. OPA (1996) refer to resources and their services (15 CFR § 990.10). Explicit references to the Economics of Ecosystems and Biodiversity (TEEB) list of ecosystem services appear in more recent implementing regulations in other countries, including for Indonesia and Brazil (Jones et al., 2015). See further discussion in Section 5.3.

[2] See Carson et al., 2003 for a description of the contingent valuation study commissioned by the State of Alaska. For a description of the natural resource injury and case settlement see: https://www.justice.gov/enrd/us-v-exxon-corporation-et-al-dalaska

and hazardous substance spills or long-term discharges. These statutes establish prevention and response policies, in addition to the restoration and compensation requirements of the liability provisions. Alternatively, the National Marine Sanctuaries Act (1988) and the Park System Resource Protection Act (1990) establish protected areas for special resources and mandate the development of resource management plans, in addition to creating liability provisions for injuries to the protected resources from any source.[3]

All of the U.S. natural resource liability statutes call on the president, state governors, and sovereign tribal nations to designate officials from natural resource management agencies to serve as trustees for natural resources on behalf of the public. The natural resource trustee concept draws on the common law principles of the public trust doctrine and *parens patriae*, whereby the sovereign has certain legal obligations to protect and preserve the trust corpus (Ward and Duffield, 1992). The trustees are granted standing to claim compensation for damages and to use the recovered monies to compensate the public by restoring, rehabilitating, replacing, or acquiring the equivalent of the injured resources. Natural resources are defined broadly to include land, fish, wildlife, biota, air, water, groundwater, drinking water supplies, and other such resources belonging to, managed by, held in trust by, appertaining to, or otherwise controlled by the United States, any state or Indian tribe, or any foreign government. [4]

The measure of damages and the allowable methods to estimate them, as interpreted in the implementing regulations for CERCLA (1986, 1987)[5] and for OPA (1996) have both been subject to legal challenges. The D.C. Circuit Court has issued precedent-setting opinions in response to several suits,[6] which strongly affirm that (1) the measure of damages is the cost of restoration of the injured resources and their services plus compensation for the interim lost value pending recovery of the resources and their services to baseline, including direct use value and passive use value; (2) trustees are not limited to valuation methods specifically identified in the regulations and do not need to provide detailed standards for the use of specific methods; and (3) the contingent valuation method may be reliable for measuring passive use value. In a continuing dispute over whether the CERCLA measure of damages covered the restoration of resources or their services or both, the Court also clarified that the first component of the damage claim–''restoration of injured resources''–refers to the cost of restoring injured resources to their *baseline level of services* (Kennecott, 88 F.3d at 1220) (Jones, 1997; Lee and Bridgen, 2014).

The court affirmations that passive use value can be included in the damage claim and that contingent valuation methods may be used for estimating damages are critical to the ability of the trustees to incorporate the value of the full range of resources and ecosystem services in their damage claims. For both CERCLA and OPA regulations, the courts are to assess the admissibility of scientific studies on a case by case basis, based on criteria including reliability and validity**.** In practice, over the past 25 years since the *Exxon Valdez* litigation, natural resource damage claims infrequently have valued passive use losses using stated preference methods, which are costly to implement rigorously and remain controversial.

Box 1 Ecosystem services and valuation: approaches and challenges in U.S. natural resource liability policy. The concept of natural resources as *natural capital that provides flows of various services of value to humans* has been reflected in the NRDA regulatory provisions from the initial regulations (1986, 1987) for Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) and further elaborated in the Oil Pollution Act (OPA) regulations (1996). The framework laid out in these regulations drew upon concepts both from the ecosystem services (ES) literature (Daily, 1997; Heywood, 1995)[7] and from the environmental economics literature on non-market valuation (NMV) (Freeman, 1993; Freeman et al., 2014; Kopp and Smith, 1993; Ward and Duffield, 1992). The two overlapping literatures have developed in parallel, with extensive cross-fertilization, but with some differences in focus (Farley, 2012; Ma and Stern, 2006).

Among the several categories of ''ecosystem services'' identified in the ES literature, the OPA guidance highlights the linkages between direct human uses (''provisioning'' and ''cultural/amenity'' services in ES literature) and the ecological services that provide inputs for direct human uses (''regulating'' and ''habitat/support'' in ES literature). Based on Daily (1997), Fig. 1 (NOAA, 1997) categorizes human uses for a wetland example in such a way as to highlight the mapping between each human use category and the available valuation methods for estimating its value, as articulated in the NMV literature (Freeman, 1993). The first five categories of human uses provide services directly to individuals (*direct uses*); the last category, passive use value,[8] captures the fact that individuals may value natural resources independent of direct uses, but rather for their own sake, for bequests to future generations, or for the services provided to others.

For valuing direct uses, revealed preference and production function (or its dual, factor income) methods are available. Estimated from data on observed economic choices, revealed preference methods include models of market demand and supply to value marketed goods (including production inputs and consumer goods), travel cost to value recreational or subsistence uses, and hedonic models to value the environmental attributes of market goods and services.

Replacement cost is also sometimes listed as a valuation approach, but is only consistent with valuation principles if the replacement actually occurs and the injured party is thereby made as well off as before the injury; otherwise, using cost as a proxy for value may understate or overstate value. In addition, stated preference methods, such as contingent valuation, contingent behavior, and discrete choice experiment methods (which include conjoint analysis), are available for valuing direct uses. For passive uses of resources, there is—by definition—no observed choice behavior to provide the data for revealed preference methods: consequently, only stated preference methods are available to value passive uses.

*(continued on next page)*

[3] Other U.S. federal statutes containing natural resource trustee provisions include the Deepwater Port Act of 1974; Outer Continental Shelf Lands Act Amendment of 1978; and the Trans-Alaska Pipeline Authorization Act, 1973.

[4] This language is the same in the CERCLA definition (42 U.S.C. §9607(f)(1)(2)) and the OPA definition (33 U.S.C. § 2701(20)).

[5] They also applied to Clean Water Act cases before promulgation of OPA.

[6] The precedents were initially set in *Ohio, 1989*, the litigation over the original regulations implementing CERCLA, promulgated by US DOI, and reaffirmed in subsequent decisions (*Kennecott 1996* re: the CERCLA rules and *GE 1997* re: the OPA rules).

[7] The ES literature taxonomy of services, both for the individual services and the categories used, has evolved over the past two to three decades since the regulations were promulgated, but has remained fairly consistent. See De Groot et al., 2010 for a comparison of lists over time.

[8] The ES literature has alternatively used the term ''non-use'' value (De Groot et al., 2010). The NMV literature shifted away from this usage in the 1990s, on the grounds that it does represent a use—just not a direct use (Freeman, 1993).

*(Box1 continued)*

Quantifying the relationships between ecological services and human uses and values—as illustrated in the iconic diagram in Fig. 2—is challenging: whereas measurement of biophysical structure and processes and ecological functions may be feasible, direct measurement of the ecological services they provide typically is not feasible. For contexts where the contribution of ecological services to direct human uses (such as wetlands protecting drinking water quality, or protecting coastal property and infrastructure from storm surges) is clear, it may be possible to estimate production function models to characterize those relationships. To value the (changes in) human uses projected by such a model, the standard strategy is to transfer use values published in the literature for another location to the current context. Such "benefits transfer" can be done at substantially lower cost than to collect site-specific data for estimating site-specific values from revealed or stated preference methods, though the suitability of the available "benefits transfer" estimates to the analysis context varies. Applications of this production-function approach to ecological services were very limited when the CERCLA and OPA regulations were written. An exception was the "Type A model", developed in response to a CERCLA statutory mandate, for use in small spills of oils or chemicals in marine and coastal environments and in the Great Lakes (in a second model version). Originated to inform valuation of monetary damages, the model provided an integrated approach for measuring injuries and damages, with sub-models capturing physical fates, biological effects on key habitats and species, and economic damages associated with lost habitat and commercial and recreational uses (French, 1996; French-McCay, 2004; Grigalunas et al., 1987).

Over the past 15 years, though, substantial resources have been invested in developing such ecosystem modeling tools, and they have been applied in a variety of ex ante planning and management contexts. Nonetheless, substantial challenges remain to data collection and modeling, particularly to capture services and values at a fine scale (De Groot et al., 2010; Guerry et al., 2015; Ruckelshaus et al., 2015).

### 2.2. Monetary and resource compensation illustrated

The OPA measure of natural resource damages includes (1) the costs of restoring, rehabilitating, replacing, or acquiring the equivalent of the damaged natural resources ("primary restoration") and (2) the diminution in value of those natural resources pending recovery of the resources to baseline, but for the injury ("compensation for interim losses").

Primary restoration projects may expedite and/or increase the likelihood of recovery; consequently, the benefits of projects accrue as reductions in the interim lost value experienced by the public due to resource injuries. Consider a hypothetical incident where an oil spill at time $t_0$ impairs the ecosystem services provided by a wetland. Fig. 3 illustrates how the choice to implement a primary restoration project (for example, revegetating the oiled wetland) will reduce interim losses relative to natural recovery. The horizontal axis represents time, and the vertical axis represents an indicator variable of an ecosystem service (or an index of multiple ecosystem services) provided by the wetland. With natural recovery, the injury site is projected to return to baseline service levels at time $t_2$. However, conducting active restoration is projected to expedite recovery, such that service levels return to baseline by time $t_1$ and interim losses decline from area $(A_1 + A_2)$ in the natural recovery scenario to area $A_1$.

In the original CERCLA regulations (1986, 1987), the economic measure of damages for the interim loss of resources was the interim lost value incurred by individual members of the public, measured in money terms ("monetary compensation") (43 C.F.R. § 11.80 b)). For the case illustrated in Fig. 3, the damage claim with natural recovery would be the present discounted value of area $A$ ($=A_1 + A_2$); with primary restoration, the damage claim would be the cost of the primary restoration actions *plus* the present discounted value of area $A_1$. This measure of damages is well suited for litigation where individual claimants receive monetary compensation directly—for example, in private tort suits—because it indicates the amount of money that will make the affected individuals as well off as they were before the incident.

With the resource compensation approach, the measure of damages for interim lost services is the cost of the preferred compensatory restoration project(s). In Fig. 3, the compensatory restoration project is initiated sometime after primary restoration was initiated. To ensure that the public is fully compensated for the interim losses, a scaling process determines the size of compensatory restoration actions necessary for the present discounted value of the gains from the actions (Area $B$) to equal the present discounted value of the interim losses (Area $A_1$). In some cases, the injured site is sufficiently degraded before the incident that it can be rehabilitated to provide ecological functions beyond the pre-spill baseline level. Alternatively, the compensatory restoration project may be performed at another site nearby.

### 2.3. Preparing a restoration plan under the OPA regulations

In the OPA regulations, several stages are involved in development of a restoration-based damage claim (the "restoration plan"). The first step is to determine and quantify the injuries to natural resources and/or services. Then, the elements of the restoration plan include (1) identifying multiple restoration alternatives, including primary and compensatory restoration actions, (2) scaling each alternative so that compensatory restoration compensates for interim losses, and (3) selecting a preferred restoration alternative. To provide the context for our more in-depth discussion of scaling approaches in Sections 3 and 4, we outline the activities and decision making criteria for the injury quantification and restoration planning phases, and we highlight how the concept of ecosystem services serves as an organizing principle in the criteria for decision making for each.

#### 2.3.1. Determining and quantifying the injury to natural resources and/or natural resource services

The goal of injury assessment is to determine the nature, degree, and extent of any injuries to natural resources and services. In OPA, injury is defined as "an observable or measurable adverse change in a natural resource or impairment of a natural resource service" (15 CFR § 990.30).[9] Injuries are quantified by comparing the condition of the affected natural resources and their services – including degree and spatial and temporal extent of injuries – to their baseline condition, which refers to the conditions of the resources and services that would have existed if the discharge had not occurred (15 CFR § 990.52). This information is necessary to provide a technical basis for evaluating the need for restoration actions, as well as their essential characteristics and scale.

[9] In the earlier CERCLA statute, the injury definition does not make reference to ecosystem services: *injury* is defined as a "measurable adverse change, either long- or short-term, in the chemical or physical quality or the viability of a natural resource resulting either directly or indirectly from exposure to a discharge of oil or release of a hazardous substance, or exposure to a product of reactions resulting from the discharge of oil or release of a hazardous substance." Further, "injury encompasses the phrases 'injury,' 'destruction,' and 'loss' " (43 CFR § 11.14(v)).

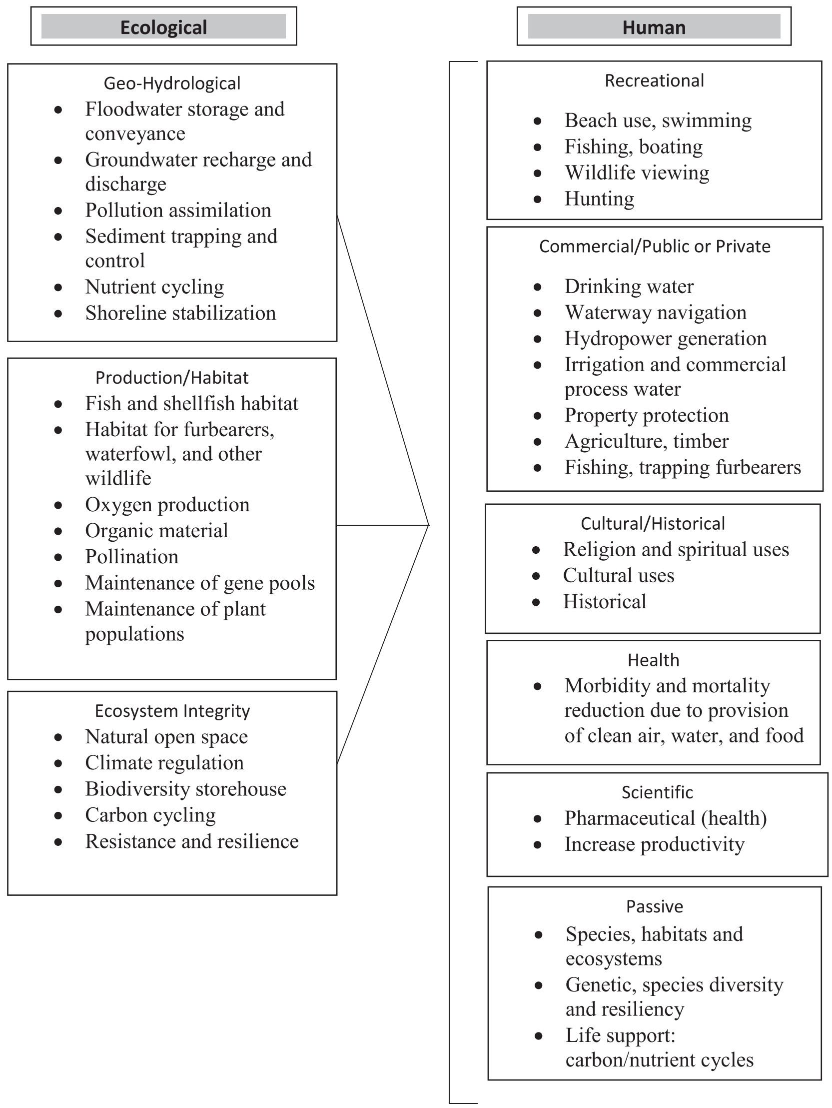


**Fig. 1.** Wetland ecosystem services and sources of value. Source: NOAA, 1997, pp. 2–3.

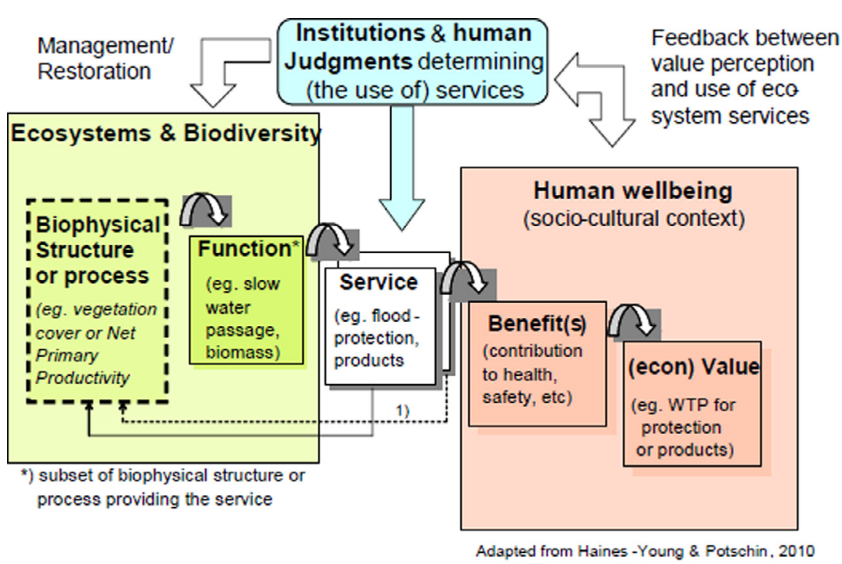


**Fig. 2.** The pathway from ecosystem structure and processes to human well-being. Source: (De Groot et al., 2010, p. 11). (See above-mentioned references for further information.)

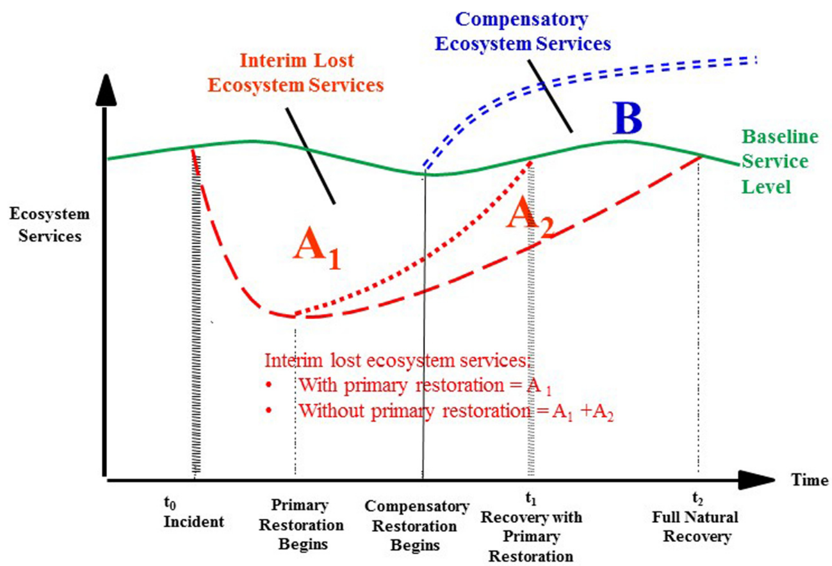


**Fig. 3.** Measure of damages with alternative forms of compensation for interim losses. The full damage claim includes (1) the cost of primary restoration (if any) *plus* (2) ***monetary*** or ***resource*** compensation for interim losses, where monetary compensation for interim losses = PDV($A$); resource compensation for interim losses = cost of compensatory restoration project for which [PDV($A$) = PDV($B$)]; and where $A = A_1$ if primary restoration occurs and $A = A_1 + A_2$ if not. [PDV refers to "present discounted value"].

For reasons of feasibility and cost, the strategy is to select a limited, but representative, subset of habitats and species that are important ecological resources providing significant ecosystem processes, functions, services, and ecological linkages that are relevant to restoration planning. The representative habitats and species are selected based on data and information gathered during the pre-assessment phase and in consultation with local and federal agency resource scientists with relevant expertise. The data collected are then used to develop scientifically based conclusions about type and degree of injury to the specific areas studied, and also—by scientific inference—to resources and ecological processes that could not be studied directly.

For large and complex NRDAs, this strategy for injury quantification may result in numerous scientific studies that include sampling for multiple metrics for each injury studied. For example, the U.S. conducted its largest and most comprehensive assessment ever for the 2010 *Deepwater Horizon* incident, which was the largest accidental offshore oil spill in the U.S. (Deepwater Horizon Natural Resource Trustees, 2016). For more typical cases, the trustees select a single or small number of metrics to monitor—typically of ecological processes or functions—as indicators of key categories of ecosystem services, based on established relationships between the metrics and the ecological services they represent, as we discuss further in Section 3.2.

### 2.3.2. Developing a restoration plan

*2.3.2.1. Identifying restoration actions: criteria for selection.* Once the trustees have identified and quantified the impaired resources and ecological processes, functions, and/or services, the next step is to identify primary and compensatory restoration actions.

**Primary restoration**—which encompasses "restoration, rehabilitation, replacement, or acquiring the equivalent of the injured natural resources"—is intended to increase the likelihood and/or speed of the recovery of the injured resources. Classes of primary restoration actions include reconstructing physical habitat that was destroyed, taking measures to protect or increase the population of an affected species, and taking actions to remove residual sources of contamination that were not addressed under OPA or CERCLA response and remediation. NOAA's guidance document on primary restoration documents prior experience with each type of action for a wide range of habitats and resources, including assessments of feasibility, constraints to implementation, and documented success rate for each approach (NOAA, 1996). With the expanding restoration activities providing field sites for research in the intervening decades, the field of restoration ecology has made substantial advances since that time (Perring et al., 2015).

For example, primary restoration actions identified for salt marshes include natural recovery, replanting, supplementary erosion control structures if necessary to stabilize the marsh sediment, contaminated sediment removal/replacement, vegetation cropping to remove contaminants, new saltmarsh creation, low pressure flushing and bioremediation (NOAA, 1996, pp. 2-7–2-23). The literature review highlights the injury that can occur to the soft sediments of salt marshes with improper restoration, and aggressive efforts at removing contamination through flushing or removals.

In some instances, the injury is not expected to recover through natural processes and restoration or rehabilitation of the injured resources and services may be counterproductive, infeasible or cost-prohibitive. In this case, restoration will involve *replacement or acquiring the equivalent of the injured resources and services*. Further, the analysis of the replacement option effectively becomes part of compensatory restoration scaling for interim losses pending recovery to baseline. The scaling for compensatory restoration will take into account that interim losses will continue in perpetuity.

**Compensatory restoration** is intended to compensate the public for interim losses. Classes of compensatory restoration actions include enhancing impaired resources and providing replacement resources that are equivalent in quality and value to the injured resources. For example, for contaminated wetland habitat, projects could include rehabilitating degraded wetlands or acquiring and protecting wetlands threatened by development. For impaired resources (such as fish or birds), options include rehabilitating animals, enhancing spawning, nesting and foraging habitat, managing predators, re-establishing breeding colonies, and reducing fish by-catch.

Replacement *in kind* is the highest priority (15 CFR § 990.53(c)(2)).[10] For example, if a wetland that provides storm protection and

[10] The regulations state: "To the extent practicable, when evaluating compensatory restoration actions, trustees must consider compensatory restoration actions that provide services of the same type and quality, and of comparable value as those injured." (15 CFR § 990.53(c)(2).

water quality services for watershed communities is injured, then the highest priority for compensatory restoration is to provide a wetland project providing those same services (i.e. the "same type of resource and ecosystem services"). In recognition that the long history of restoration for mitigation is replete with examples where the restoration projects of the same habitat type have not provided ecological function or services comparable to lost habitat, OPA regulations further highlight that compensatory restoration actions that provide resources and ecosystem services of the same type should also provide ecosystem services of the *same quality* and of *comparable value* as those injured (Id.)

Drawing on lessons from the scientific literature on wetlands mitigation (King and Adler, 1991), OPA guidance directs the trustees to look beyond the type of habitat at a particular replacement site to assess the quality and value of ecosystem services the site will provide, by focusing attention on the capacity, opportunity and payoff (i.e. benefits) for the site to provide ecosystem services comparable to the injury site (NOAA, 1997). On-site biophysical characteristics (e.g. soil, vegetative cover, hydrology and faunal density) affect the capacity of an ecosystem to provide storm protection and habitat-related recreational services. Landscape context affects whether the ecosystem will have the opportunity to supply many of the ecological and human services and strongly influences whether humans will value the opportunities for services. For example, a wetland's capacity to provide sediment trapping depends on such factors as slope and vegetative cover. The opportunity for the wetland to trap sediments depends on the expected flow of sediments through the wetland, which will depend upon types of upland land uses (i.e. landscape context). The total value generated from water quality improvements due to sediment trapping will depend upon the uses of the affected downstream water bodies: the value will be greater if there are nearby shellfish beds and finfish spawning areas than if the water flows into a fast-moving river.

In addition, implicit in the regulations and explicit in the guidance document is a preference for siting projects—to the extent feasible—such that they benefit the members of the public harmed by the incident.

In some cases, replacement with the same type of resources and services is not feasible or cost-effective, and substitutions will be required. The next priority is to assess whether it is possible to provide comparable services with a different set of resources. For example, in the Fox River/Green Bay CERCLA site discussed below, hazardous contamination was pervasive throughout Green Bay and further remediation beyond EPA's cleanup of the site was cost-prohibitive. Because the sediments and therefore the fishery resources were going to remain contaminated for an extended time period, the trustees are implementing restoration projects providing wetlands and associated upland habitat; aquatic, nearshore and riparian habitat quality improvement; fishery resource enhancements, and improved natural-resource-based public access enhancement (where the last is limited to less than 10% of total restoration funds) (Fox River/Green Bay National Resource Trustee Council, 2016; Stratus Consulting, Inc., 2013).

*2.3.2.2. Scaling restoration alternatives: criteria for choice of approach and method.* For scaling compensatory restoration, the OPA regulations identify two basic approaches. The service-to-service approach—a simplified in-kind-trading-like approach that does not require valuation—is applicable for resources and services provided by compensatory restoration projects that are of the same type and quality and comparable value as those injured. The more general valuation approach relies on survey-based stated preference methods to value the tradeoffs between injuries and prospective compensatory restoration projects, particularly in the case of indirect or passive human uses.

The choice of a scaling approach cannot be made without identifying the data and methods that are available for implementing the scaling approaches under consideration. The integrated choice of scaling approach and methodology depends upon an evaluation of three sets of criteria (OPA §990.27):

- *Applicability* of the approach and methods in the particular context;
- *Reasonableness of the incremental costs* of a more complex approach and/or methods, relative to the expected increase in the quantity and/or quality of relevant information and
- *Validity* and *reliability* of the approach and the methods to implement it in the particular context.

For injuries to the difficult-to-value resources and ecological services that provide indirect or passive human uses, we found that trustees have relied almost exclusively on HEA, a service-to-service approach, including for contexts where trustees make resource substitutions, which we review extensively in Section 3 below. The other most frequent category of ecosystem services losses are lost direct uses for recreation. We found that for this class of losses (which are outside the focus of the paper), the travel cost model, relying on observed behavior data, and in some cases stated behavior or stated preference data, were employed. We discuss this pattern briefly in Section 4 below.

*2.3.2.3. Selecting a preferred alternative: criteria for selection.* In their restoration plan, which is to be submitted for public review and comment, the trustees evaluate a set of restoration alternatives and select a preferred alternative. One alternative is to include natural recovery as the primary restoration action, which generally leads to a higher level of interim losses and therefore a larger scale of compensatory restoration actions.

The evaluation criteria for selecting the preferred alternative include the anticipated efficacy in returning the injured natural resources and services to baseline and/or compensating for interim losses and avoidance of collateral injury to resources or public health and safety. Restoration costs are also a criterion for evaluation, and a cost-effectiveness test is to be applied if two or more alternatives are equally preferable. However, the trustees are *not* required to develop the data necessary to demonstrate that the benefits of the preferred alternative pass a benefit-cost test.

## 3. Scaling compensatory restoration: service-to-service approach

Service-to-service scaling is analogous to an in-kind trading approach, which does not require explicit valuation of interim losses or of restoration project gains. Use of the approach presumes an in-kind equivalency between ecosystem services lost through injury and those gained through restoration projects. With equivalent value per unit of services lost and gained, the scaling analysis simplifies to selecting the scale of a restoration action for which the present discounted quantity of replacement services equals the present discounted quantity of services lost due to the injury. Scientific inferences and judgments are required to choose metrics that identify equivalencies for any substitutions across landscape, time and habitat/species; these judgments are analogous to those required for various environmental trading programs, including wetlands and conservation mitigation banks.

### 3.1. Habitat equivalency analysis

#### 3.1.1. Overview

HEA is the most common service-to-service approach used to scale identified compensatory restoration projects. It simplifies complex ecosystems through the choice of representative ecological process or function metrics as proxies for the change in the quantity and quality of service levels at the injury and restoration sites in a particular case. Following the reasonable cost criteria for selecting assessment methods, the depth of the analysis will be determined in part by the scale and significance of the injuries.

NOAA initially used HEA to calculate NRDA damage claims in the early 1990s for ship grounding cases that destroyed small quantities of protected coral reef and sea grass in the Florida Keys National Marine Sanctuary (FKNMS) (NOAA, 1995). Due to HEA's simplicity and flexibility in addressing hard-to-value ecological injuries, trustees have subsequently used HEA for cases involving oil spills, simple and complex hazardous materials releases and forest fires, and for injuries affecting a wide range of habitats (including wetland vegetation, rivers and riparian habitats, beaches, coral reefs, subaquatic vegetation habitats and benthic habitats) and resources (including fish, aquatic invertebrates and birds, particularly for endangered species). Resource equivalency analysis (REA) and landscape equivalency analysis (LEA) are variations on the HEA approach that are used to scale the injury when it involves primarily one or more natural resource species, rather than habitat.[11] For simplicity, we will use the term HEA to cover all of the variations.

HEA has proven to be a highly successful tool for achieving restoration settlements, which is how most cases are resolved in the U.S. The HEA approach was upheld by the court in the few federal cases that were litigated to conclusion. In two early cases, federal courts awarded damages for the destruction of 1.6 acres of sea grass during 1992 treasure hunting activities in the FKNMS (*U.S. v Fisher* et al., 1997, 1999) and for the destruction of 2.1 acres of sea grass as a result of dragging dredging equipment through sea bottom habitats of turtle grass, manatee grass, and finger coral (*U.S. v. Great Lakes Dredge and Dock Co.* et al., 1999, 2001). More recently, federal courts upheld the use of HEA for valuing fire damage to forest lands owned by the U.S. Forest Service in 2008 and 2012 (*United States v. Union Pac. R.R. Co.*, 2008 and *United States v. CB & I Constructors, Inc.*, 2012).

Various economists initially outlined the conceptual foundation for the in-kind trading logic used in HEA (King and Adler, 1991; Unsworth and Bishop, 1994). Jones and Pease (1997) offered a welfare economics foundation for the concept of resource compensation set out in the OPA regulations, and the use of HEA to quantify it. After some discussion in the literature about what simplifying consumer preference assumptions are necessary for the HEA framework to provide utility-theoretic compensation (Dunford et al., 2004; Flores and Thacher, 2002; Zafonte and Hampton, 2007), Zafonte and Hampton concluded that the traditional HEA framework provides an acceptable approximation of aggregate compensation for a reasonably wide range of consumer preference and ecological parameter combinations.

As developed in Jones and Pease (1997), the HEA formula can be written

$$\sum_n \sum_{t=0}^{B} v_n * \left[(\text{loss in services/acre})_t/(1+r)^t\right] * J \text{ acres of habitat injured} \approx \sum_n \sum_{t=I}^{L} w_n * \left[(\text{gain in services/acre})_t/(1+r)^t\right] * C \text{ acres of replacement habitat} \tag{1}$$

where $v_n$ is the value per service-acre-year of injured habitat to individual $n$, $w_n$ is the value per service-acre-year of replacement habitat to individual $n$, $r$ is the real rate of discount, the habitat is injured in year $t = 0$ and recovers to baseline in year $t = B$, and the increment in replacement services begins in year $t = I$ and ends in year $t = L$.

*3.1.2. Implementation*

Under simplifying conditions, the value terms cancel out of both sides of the equation, and the equation then can be solved for the scale of the compensatory restoration action, $C$ acres of the identified replacement habitat. As laid out in the regulations, the conditions are:

(i) a common metric can be defined for services that captures any significant differences in the quantities and *qualities* (italics added) of services provided by injury and replacement resources,
(ii) the changes in resources and services from the injuries and restoration actions are sufficiently small that the value per unit of service is independent of the changes in service levels, and
(iii) the extent of the "market" (i.e. those individuals who value the resources) is the same for injury and replacement resources.

The first condition, which pertains to defining an effective metric to proxy for services, implicitly embodies two assumptions: (1) the chosen metric—whether a single metric or an index aggregating multiple metrics—can adequately capture the range of ecosystem functions and (2) service levels occur in the same proportion to the chosen ecological process or function metric at both the injury and restoration sites. The second condition pertains to the comparability of value of ecosystem services at both the injury and restoration sites. Barbier (2011) suggested this assumption could be problematic because the preference for resource replacements of the same type and in the vicinity of the harm could lead to an "excess supply"—say of replacement riparian wetlands—that provides limited additional value. Other authors have highlighted the implicit assumption that values are fixed over time, given that the injury and restoration services occur during different time periods (Dunford et al., 2004). The third condition highlights equity issues in locating the project so that those injured are beneficiaries of the replacement resources.[12]

As an example, we draw on the wetland oil spill illustrated in Fig. 3, where the baseline level of services at the injury site is represented by the solid green line. If the primary restoration project revegetating the oiled wetland is implemented, the injury recovery function is the dotted red line outlining area $A_1$. For compensatory restoration, which involves rehabilitating a nearby degraded wetland area, the augmentation of services is represented by the difference between the double-dashed (blue) line for replacement project services and the baseline level of services, the solid (green)

[11] In resource equivalency analysis (REA), the injury typically is measured in terms of number of individuals killed or loss of reproductive capacity. Restoration is scaled to provide a resource replacement equivalent to the estimate discounted lost organism years, or habitat or habitat improvements that will allow the species to reproduce naturally in sufficient numbers to compensate for the lost organism years (Strange et al., 2004). Landscape equivalency analysis (LEA) has been proposed as an extension of HEA and REA for conservation banking, particularly for projects to protect or restore endangered species; it adds quantitative indicators of spatial and population genetic aspects into the valuation of habitats and species (Bruggeman et al., 2005).

[12] Kopp and Smith (1993, p. 327) highlighted the role of the spatial range of resource users relative to the resource, drawing a parallel to the requirements to delineate the extent of product markets for consideration in mergers and acquisition cases.

line. To achieve comparability, the service metric must be in the same units.

The parameters required for implementation include those for injured areas, replacement areas, and time equivalency.

*Injured area parameters:*

- Baseline level of services at the injury site, which is normalized to be 100% service-acres;
- Initial extent and nature of the injury: the spatial extent of injury (in acres, for example) and the initial loss in service level relative to baseline at the injured site (characterized as a percent of the baseline level of services).
- Injury recovery function (with primary restoration or natural recovery), including functional form, maximum level of services to be achieved (characterized as a percent of the baseline level of services) and recovery period for injured resources.

*Replacement area parameters:*

- Initial level of services at the replacement project site, characterized in the same unit of services as for the injury area and
- Replacement project service level function, including functional form for (incremental) service maturation, the maximum level of services, service maturation period, and replacement or creation project duration.

We normalize service levels at the injury and replacement sites by dividing by the baseline service level at the injured site and translating into percentages; this way, the projected baseline service level is set at 100% in each time period.[13]

*Time-equivalency parameter:*

- Annual real discount rate.

The injury losses and restoration gains are measured in discounted service-acre years.

#### 3.1.3. Key factors for effective compensation

Two factors are key to scaling compensation to provide equivalent value to the interim losses: the choice of restoration project and the choice of the service metric, along with other parameters in the HEA. First, the selection of restoration projects and sites is critical to achieving comparability of *quality and value of services* at injury and restoration sites. Taking into account considerations of capacity, opportunity, and payoff for ecological services, as well as the equity of the payoff, is paramount to ensuring that projects of the same type (e.g. wetlands) achieve comparable quality and value.

Second, as various authors (Barbier, 2011; Dunford et al., 2004) have pointed out, the calculation of the scale of the compensatory project is sensitive to the choice of parameters, including the service metric. The HEA calculations also must draw upon the literature on parameter values or models for injury recovery functions and restoration maturation functions. Some ecological metrics may overstate the extent to which baseline service levels are replaced or may underestimate the time required for recovery. Using salt marsh restoration as an example, when structural metrics such as biomass cover have fully recovered, ecological processes necessary for full functioning of the marshland, such as nutrient cycling, may lag significantly behind, which should be factored into determinations of overall habitat recovery rates (Strange et al., 2002).

The reality is that the depth of data collection and analysis will be determined in part by the scale of the case. To conduct a cost-effective assessment, the choice of metrics must balance the potential for the metrics to be observed with reasonable precision and at reasonable cost against their potential to account for the complexity of the affected ecological services, which typically are difficult or infeasible to measure directly. Important factors to consider when selecting metrics for application in an HEA scaling framework include

- Relationship of the metric to habitat structure, function, and ecosystem services of concern for the incident under evaluation.
- Relationship to the flow of ecosystem services of societal value.
- Sensitivity of metric to the environmental insult.
- Feasibility and cost to collect and measure injury (and restoration) data for the metric in field and/or laboratory settings. Are specialized expertise and equipment required? Has the resource been cultured and tested in a laboratory setting before?
- Considerations about the nature of sampling required for the metrics, especially considering destructive versus non-destructive methods.
- Knowledge of natural variability, seasonality, and pre-release or reference information. This is important for ensuring that comparisons are made against a baseline and that determinations can be made with a reasonable number of samples. (Also, do we have confidence that we are able to measure and quantify a change if it is occurring?)

In our discussion below, we focus on the choice of ecological metrics to proxy for service flows and for estimating percent loss from injury or gain from restoration in a range of different habitat and resource contexts.[14]

### 3.2. Choice of ecosystem services metrics

We first consider the use of ecological indicators in the Blackbird Mine case, which represents one of the larger trustee settlements for hazardous waste contamination to date. (See the Appendix for more details about the case). We then summarize the procedures used in a range of habitats in different cases.

#### 3.2.1. Blackbird Mine case study: hazardous substance injury to threatened salmon populations

Contamination from the Blackbird copper mine contributed to the decline of resident trout species and the ultimate elimination of threatened spring and summer Chinook salmon and steelhead from Panther Creek in Idaho. The creek is an important source of Chinook salmon in the Salmon River, which has been classified as threatened under the Endangered Species Act since 1992. Operation of the mine also resulted in fishery habitat degradation and loss through the construction of access roads, timber clearing, sedimentation, and creek channel re-alignments. The trustees agreed that a cost-effective approach to quantify the injury—and thereby to identify the restoration goals—would be to focus on three primary indicators of the Panther Creek ecosystem: exceedance of surface water criteria; injury to streambed food web species; and injury to resident and anadromous fish (i.e. salmon that spawn in freshwater and then migrate to the sea).

Taking into account various other factors contributing to the decline in the salmon populations, trustee studies estimated that the system would be able to support approximately 200 naturally spawning adults returning to Panther Creek each year. The goals of

[13] For example if the service metric proxy is above and below ground biomass, then projected biomass levels at the injury and replacement sites will be divided by the baseline level of biomass.

[14] The literature has also featured extensive discussion of the choice of the discount rate, which is beyond the scope of this paper. See Shaw and Wlodarz (2013) for a recent review.

the restoration plans were to restore the salmon population by populating the affected creek with hatchery-sourced fish from genetically wild stock (for primary restoration) and to protect and restore fishery habitat for compensatory restoration. HEA was used to calculate the amount of fishery habitat restoration that would be sufficient to compensate the public for the interim losses.

The chosen indicator of restoration plan success was the extent to which the level of naturally spawning spring/summer Chinook salmon reaches the carrying capacity of the creek—and this was chosen as the 100% service level in HEA. Due to the threatened status of the Salmon River Chinook salmon, this ecological function measure is significant; furthermore, it conforms to the three key assumptions for equivalency identified above. The presence of naturally spawning salmon at the carrying capacity is an indicator that other restoration goals have been met. The population recovery will require compliance with the water quality standards, and the other injured resources in the aquatic ecosystem are forecast to recover on their own shortly after water quality restoration (unlike the salmon). The return of naturally spawning salmon will supply the ecological functions previously provided when the river ecosystem was populated by salmon, including return of nutrients to the relatively nutrient-poor headwaters of the streams, which provides a nutrient base for the stream. The return of nutrients will stimulate primary production, promote growth of aquatic plants, macroinvertebrates, small fish etc. and provide energy for the stream to support another generation of salmon. When the endangered status for salmon is lifted, the fish population will support recreational, subsistence, and cultural uses of the fishery (closed since 1957).

Further, primary and compensatory restoration actions are occurring in the same location as the injury, and these projects are small relative to the watershed-wide challenges that need to be resolved before the endangered status is lifted for Salmon River salmon; therefore, it is reasonable to assume that the values of the lost and gained services are comparable—excess supply is not an issue—and that the geographic extent of beneficiaries is comparable.

### *3.2.2. Other habitats and resources of the same type, quality and comparable value*

In Blackbird Mine, the ecological metric was recovery of an indicator species of endangered status. Table 1 summarizes the choices for ecological metrics and the procedures for calculating percent loss. It draws extensively from a detailed guidance document on compensatory restoration (English et al., 2009), with additional information drawn from case documents from individual cases reported on trustee agency websites.[15]

The most straightforward applications are for physical destruction. Vessel groundings in the FKNMS and elsewhere in the U.S. have provided extensive experience with injury assessment and restoration of sea grass beds and coral reefs, including some of the first applications of HEA. Though most incidents are relatively small, the approximately 650 groundings per year in FKNMS cause substantial cumulative impact. As a result, NOAA has developed assessment and restoration planning protocols for quantifying injuries and projecting injury recovery times and restoration project maturation times, building from NOAA-funded research and including retrospective studies monitoring the performance of vessel grounding sites.[16] For corals, the metric used as a proxy for service levels is three-dimensional habitat complexity (density and biodiversity). Restoration monitoring compares biological conditions on the restoration armor unit (providing a surface conducive for new coral growth) with those in the reference areas, in terms of coral density and biodiversity (Hudson et al., 2008). Choice of locations for restoration projects is informed by research assessing the relationship between place-based characteristics and recovery performance for past restoration activities. It is reasonable to assume services per unit of ecological function will be comparable at the injury and restoration sites. For sea grass beds, above- and below-ground biomass serves well as a proxy for nutrient cycling, sediment stabilization and food and shelter ecological functions provided by sea grass beds (Kirsch, 2005).

Marsh habitat is one of the most frequent habitats impaired by oil spills. To represent marsh habitat loss, the simplifying choice to use a single or a few metrics to represent the degree of impairment of ecosystem services is based on past research establishing the ecological relationships between degree of marsh oiling and lost ecological functions. For example, in the Chalk Point (Maryland) oil spill assessment, the trustees measured and compared vegetative metrics (such as stem height and stem density), sediment chemistry and abundance and composition of marsh fauna in relation to degree of marsh oiling (e.g. lightly, moderately or heavily oiled) (Chalk Point Natural Resource Trustees, 2002). In this case, the above-ground vegetation metrics were representative of the range of ecological functions and services related to primary production, habitat structure, recreational and aesthetic value, food chain support and fish and shellfish production. Determination of soil function was evaluated separately from the marsh, but was considered together when determining the overall loss of marsh services associated with oiling.

Fish and shellfish mortality determinations may be based on field surveys documenting and quantifying dead animals and then extrapolating to the affected area (e.g. the Alafia River (Florida) spill); alternatively, they may be based on a combination of field observations of dead animals in combination with modeling estimates of oil and chemical fate and biological effects using models (e.g. Spill Impact Model Application Package or SIMAP) (French McCay et al., 2003).

### *3.2.3. Habitat and resources not of same type, quality and comparable value*

In some cases, replacement with the same type of resources and services is not feasible or cost-effective, and the trustees make substitutions which requires cross-habitat tradeoff determinations. For example, it is difficult to restore some low-productivity habitats (such as benthic habitat); replacement with higher-productivity, higher-energy habitats is a more cost-effective use of restoration funds. English et al. (2009, p. 89) note that high-value habitats are those most seriously depleted and in most need to restoration, which motivates trustees to select salt marsh, oyster reef, mangrove forest coral reef or seagrass for habitat restoration to compensate for injuries to unvegetated sedimentary bottom. For example, in the Hylebos waterway assessment, the trustees weighted Chinook salmon habitat higher than habitat for English sole due to Chinook salmon's status as a threatened species under the Endangered Species Act (Commencement Bay Natural Resource Trustees, 1991; Munns et al., 2009). In the Lavaca Bay NRDA, the trustees chose to implement marsh restoration projects to compensate for losses to benthic habitat, based on the opinion of resource experts, who also provided the basis for the habitat tradeoff ratios (Lavaca Bay Natural Resource Trustees, 2001).

The tradeoff ratios for making such substitutions are constructed to reflect differences in the quality or value of the ecological functions and services, analogous to those constructed for wetlands mitigation programs (Bruggeman and Jones, 2008;

[15] The two major federal trustee websites are NOAA (https://www.darrp.noaa.gov/explore-projects) accessed July 29, 2016 and USDOI (https://www.doi.gov/restoration) accessed July 29, 2016.

[16] For example, a study field-validating the cellular automata model used to forecast sea grass recovery rates in NRDA cases at 30 documented vessel grounding sites found that the recovery model was forecasting a much faster recovery than observed to occur, particularly for the dominant species *Thalassia testudinum* (Uhrin et al., 2009).

**Table 1**
Choice of restoration project type and ecosystem service proxy metrics for scaling compensatory restoration projects.

| Habitat Type | Source, type of injury | Measurements to quantify injury | Metrics/approach to estimate% service loss | Type of restoration project | NRDA incident | Publications |
|---|---|---|---|---|---|---|
| Seagrass | Physical destruction of plants from vessel groundings | Seagrass short shoot counts and density | Loss of above and below ground biomass (from literature-based relationship with shoot densities) | Created new seagrass beds | US v. Great Lakes Dredge and Dock Co. (2001), US v. Fisher (1999) | Fonseca et al., 2000; Uhrin et al., 2009; Kirsch, 2005 |
| Coral | Physical destruction of coral from vessel groundings | Measurement of loss of percent cover, abundance and vertical structure of corals and other large epibiotic species | Declines in percent cover and abundance | Created new coral reefs | US v. M/V Miss Beholden (1995) | Hudson et al., 2008; Julius et al., 1995 |
| Marsh | Oil spill | a) Vegetation: coverage and thickness of vegetation oiling, stem height, density, percent cover; b) Soils: oil concentrations on surface and subsurface soils, abundance and composition of benthic macroinfauna | Averaged % service loss for vegetation and soils inferred as a function of degree of oiling | Create tidal marsh | Chalk Point MD oil spill (2000) | Chalk Point Natural Resource Trustees, 2002 |
| Oyster Reef | Physical destruction due to dredging of contaminated area | Acres of oyster reef habitat removed | Acres of reef habitat removed from dredging (100% service losses in perpetuity) | Created new oyster reef | Lavaca Bay TX hazardous waste site | Lavaca Bay Natural Resource Trustees 2001 |
| Oyster Reef | Hazardous substance contamination | Literature based estimates of injuries based on contaminant concentrations in sediments, modeled into oyster and reef fish tissues associated with adverse effects on behavior, growth, reproduction, or mortality | % service losses inferred from concentration of contaminants and degree of negative biological response | Created new oyster reef | Lavaca Bay TX hazardous waste site | Lavaca Bay Natural Resource Trustees 2001 |
| Mangrove | Oil spill | Adult tree mortality, limb loss, foliage decline, growth of early life stages, changes in species composition and changes in faunal use associated with degree and persistence of oiling on and buried in sediments. | % service losses inferred from degree of oiling and degree of negative biological response | Land acquisition with active salt marsh as successional precursor to mangrove forest restoration | Tampa Bay oil spill (1993) | Mauseth et al., 2001 |
| Soft Bottom Benthos | Hazardous substance contamination | Determination of relationship between contaminant concentrations and biochemical, behavioral, growth, reproduction, mortality, population and community changes in benthic invertebrates and demersal fish, based on published data | % service losses inferred from concentration of contaminants and degree of negative biological response | Landscape ecology focused cross-resource focused projects including habitat construction, reducing contaminant loading, and creating habitat corridors | Commencement Bay hazardous waste site | Cacela et al., 2005; Wolotira, 2002 |

Sources: English et al. (2009) and sources cited in the publications column.

King and Adler, 1991; Moilanen et al., 2009). English et al. identify two basic approaches. The first approach uses energy as a metric, and converts across habitats or resources evaluated at a common tropic level (McCay and Rowe, 2003). Some support for the implicit assumption of a relationship between energy and value exists in the ecological economics literature (Costanza et al., 1997). The second approach relies more directly on professional judgment of scientists to assess relative value of the services provided by alternative habitats and sites. An advantage of this approach cited by English et al. is that multiple services can be considered jointly, and conversions do not rely on the assumption that one metric – production – is representative of all services.

In larger and more complex cases with multiple contaminants, multiple habitats and resources, and/or multiple types of resource injuries—which frequently arise in CERCLA cases—multiple metrics are aggregated to account for services losses. A variety of approaches to aggregation have been discussed in the literature (Cacela et al., 2005; Munns et al., 2009), where the conclusion is that there is no one "best" approach; rather, the aggregation approach should be tailored to the particular conditions of each case. To evaluate service losses from multiple co-occurring contaminants for the waterway assessment (part of Commencement Bay in Seattle, Washington), the trustees used a residual service loss approach for each contaminant, which was then weighted and summed to calculate a proportionally weighted total service loss (Munns et al., 2009; Wolotira, 2002).

In cooperative assessments such as for the Lavaca Bay (Texas) NRDA, the trustees and the responsible parties worked together to evaluate data about the injury (contaminant concentrations) to multiple resources (including sediments and animal tissue) by habitat and develop consensus-based injury parameters to populate the HEA (Lavaca Bay Natural Resource Trustees, 2001; Munns et al., 2009).

The restoration plans are subject to public review, empowering the public to weigh in with their opinions as to whether the proposals are acceptable—that is, consistent with their values. The public also may be consulted in the process of identifying project proposals.

## 4. Scaling compensatory restoration: valuation methods

Alternatively, the valuation approach to scaling incorporates the use of a variety of economic methods. The first version of the valuation approach in the OPA rule is referred to as "value-to-value," which involves collecting valuation data to estimate the interim loss in value from the spill, as well as the gains in value from the compensatory restoration. If the trustees judge that valuation of lost services is practical, but valuation of replacement natural resources and/or services cannot be performed within a reasonable time frame or at a reasonable cost, the regulations indicate that a "value-to-cost" version of the valuation approach alternatively may be used (§ 990.53 (d)(3)(ii)). Value-to-cost is otherwise known as monetary compensation. With this approach, the restoration plan is scaled by estimating the value (in dollar terms) of losses due to the injury and allocating that amount toward covering the costs of compensatory restoration.

### 4.1. Value-to-value scaling

During the development of the OPA regulations, various authors (Jones and Pease, 1997; Matthews et al., 1996; Mazzotta et al., 1994) envisioned that the choice experiment format of stated preference methods would be used to estimate the new measure of damages, resource compensation. This survey-based method can provide more detailed preference information than contingent valuation, another stated preference method that typically offers one or two fixed scenarios. With discrete choice experiments, survey respondents are asked to make choices among alternative "product" scenarios with varying quality, quantity, and cost attributes. (For an overview of the methods, see (Holmes et al., 2017).

Since the regulations were published, however, few applications have been carried out for NRDA cases.[17] The premier example to date is the Lower Fox River/Green Bay NRDA. Releases of PCBs to the Lower Fox River/Green Bay by pulp and paper companies and associated waste treatment facilities injured surface water, sediment, fish, wildlife and their supporting ecosystems and cultural resources of the Indian tribes of the area. Direct human services that were impaired include recreational and cultural uses (and commercial uses outside the scope of the NRDA). The U.S. EPA remedial action at the site would still leave enough PCB contamination of the sediments that recovery of various fish and bird species would take 100 years. The trustees evaluated removing additional contaminated sediment to expedite recovery as a primary restoration action, but ruled it out as inappropriate because it didn't meet the feasibility or cost-effectiveness criteria (the estimated cost was US$111 billion) (Wisconsin Department of Natural Resources et al., 2003, pp. 3–5). Because restoring the impaired fish in the contaminated waterway was considered inappropriate, the trustees directed their attention to restoration projects of a different type. A stated preference survey was conducted to determine public preferences for a range of restoration options designed to provide services to compensate for the injuries: remediating wetlands near Green Bay, reducing pollutant runoff to improve water quality and clarity in Green Bay and enhancing existing and/or building new park facilities. The survey was structured so that the tradeoffs between the benefits of the project types and the PCB injuries could be estimated. The survey identified a strong preference for the options restoring natural resources relative to those expanding outdoor recreation facilities. It was not feasible to further capture values for differing attributes *within* specific project types because estimation uncertainty increased with the number of characteristics that differed across alternatives, starting with three characteristics (Breffle and Rowe, 2002, p. 311, ff 13). The concept was to incorporate assessment of stakeholder tradeoffs within resource project types (wetlands, aquatic, nearshore and riparian habitat, and fishery enhancements) during the restoration project planning process.[18]

Another application was to scale compensation for the recreational use losses in the Lavaca Bay NRDA (Lavaca Bay Natural Resource Trustees, 1998). In this case, a choice experiment survey elicited respondents' intentions to visit recreational sites if site quality attributes were to be enhanced by restoration actions. The stated preference survey data were then linked with revealed preference data collected on past patterns of recreational participation.

We did not find any subsequent applications of conjoint analysis for scaling recreational projects.

### 4.2. Value-to-cost scaling

The regulations specify that, to apply value-to-cost scaling, the trustees must judge that the valuation of the lost services is practicable, but valuation of the replacement natural resources and/or services cannot be performed within a reasonable time frame or at a reasonable cost (15 CFR §990.27(aX2)). The underlying concept is that, in cases with limited damages, it may not be reasonable to incur the costs of performing the site-specific data collection and analysis required for de novo studies. Particularly if recreational losses are at stake, "benefits transfer" of literature values for user days from previous studies (in combination with calculations of changes in use based on currently available data) may be a more reasonable cost solution. However, more studies exist for valuing losses than for valuing changes in quality attributes of recreational experiences. In recognition of this, the value-to-cost option was added to the valuation approach.

Indeed most applications of value-to-cost scaling have been for recreational loss components of claims. In numerous cases, the quantity of lost recreational uses (such as fishing, boating and shoreline use) has been estimated based on survey data and combined with user-day values from benefits transfer, as one component of the claim; typically the other component is an HEA valuation of ecological losses. Examples for oil spills include Chalk Point, Maryland (2000) (Chalk Point Natural Resource Trustees, 2002; Byrd et al., 2001), and Athos, New Jersey (2004) (Athos/Delaware Lost Use Technical Working group, 2009). In other cases, such as the Cosco Busan oil spill (2007), full travel cost models were estimated (California Department of Fish and Game et al., 2012). In the Green Bay/Lower Fox River contamination case, a combined stated, preference-revealed preference form of the travel cost model was estimated to calculate compensable damages to anglers from fish consumption advisories caused by PCB contamination (Breffle et al., 2006), complementing the stated preference value-to-value scaling approach for ecological injuries discussed above. To avoid double-counting, the fishery analysis was used to value past losses, and the ecological injury and restoration analysis was used to value current and future losses.

[17] Hoehn et al., 2010 explore methodological issues in scenario design in the context of a wetland mitigation example.

[18] The valuation of the claim involved two components: in addition to this restoration planning and cost analysis conducted using a value-to-value framework, a second component captured recreational fishing losses, which were valued using a value-to-cost approach (see Section 4.2). To avoid double-counting, the co-trustees quantified past interim loss damages in terms of lost fishing use value and quantified current and future damages with the restoration project planning analysis (Stratus Consulting, Inc., 2000, pp. 3–2).

## 5. Discussion

The natural resource trustees developed HEA, which was incorporated into the 1996 OPA NRDA regulations, to fill a critical gap in the NRDA valuation tool kit for valuing ecological services that provide indirect and/or passive uses for humans. In the previous section, we discussed two implementation issues pertaining to how ecosystem services are represented in HEA: (1) achieving equivalency in compensatory restoration projects, which relies on the use of judgment in the selection of compensatory restoration projects and of the metrics to characterize services provided by projects and by injury sites, and (2) managing the sensitivity of the results (and resulting size of the damage claim) to the parameters chosen. In this section, we discuss the critique that HEA does not provide information to value the benefits of restoration, impediments to adopting production-function-based ecosystem services models, and approaches for expanding the use of restoration-based approaches to natural resource liability in other countries.

### *5.1. Is there a need for benefits data?*

Several authors have criticized HEA on the grounds that it does not generate the information needed to conduct a benefit-cost test of restoration projects and so the process may result in directing resources to projects that do not achieve the desired goals in the most efficient way (Barbier, 2013; Dunford et al., 2004; Flores and Thacher, 2002; National Research Council, 2013). However, the appropriate targets for such criticism are the statutes, which do not include a benefit-cost criterion for restoration decision making, as affirmed by the courts (*US v. M/V Miss Beholden 1994, 1995*). Instead, as noted in Section 2.3.2, the criteria for selecting a restoration alternative include a *cost-effectiveness* requirement when there are multiple alternatives that are equally preferable. Another concern is that by requiring replacement in kind, the trustees may be supplying projects for which there is limited demand (or value): once the injured resources recover, the demand for those ecosystem services may be sated. As noted above, trustees have other sources of information about the relative value of different habitats and resources, including public participation, which is required at various stages in the design of the restoration plan. Trustees place top priority in restoration on injuries to protected species and habitats, which by definition represent over-stressed and under-supplied resources. Resource protection, management and restoration plans, whose use is encouraged for restoration planning where available, identify high value habitats that are the most seriously depleted and in most need of restoration and provide vetted lists of replacement project options.

On the other hand, requiring the collection of benefit information could impose various time and money costs on the process. For example, collecting the data to make such an assessment may be costly and therefore would significantly increase the costs of the damage assessment, which costs are part of the damage claim presented to the potentially responsible party. Furthermore, conducting such an analysis presumes that the details for the restoration projects have been decided, which could potentially delay resolution of a case and could impose a substantial burden on the responsible party. If the proposed projects are excessively costly and violate the cost-effectiveness criterion, the adversarial process in litigation (and settlement) is likely to generate restoration alternatives of lower cost for comparable services, thus triggering the cost-effectiveness requirement.

The federal advisory committee review of natural resource damage assessment procedures commissioned by USDOI in 2005 concurred that with few exceptions, "restoration of injured resources can be achieved more quickly, more efficiently, and more effectively by focusing on restoration in lieu of monetary damages, and on cooperative approaches to assessing and addressing injury" (US Department of the Interior, 2007, p. 6). They concluded that, based on more than 20 years of practical experience, "consensus-based approaches to dealing with scientific uncertainty, clear restoration-based objectives, and close coordination with a broad spectrum of environmental protection and natural resource conservation interests and authorities have proven to be the most successful strategies for resolving claims and achieving restoration" (Ibid.)

### *5.2. Can ecosystem services production-function models replace HEA?*

The U.S. National Research Council (NRC) recommended that the *Deepwater Horizon* oil spill natural resource trustees supplement traditional assessments using HEA (which they caricature as focusing on resources, without regard for the associated human services and values) by undertaking a more comprehensive ecosystem services approach to the NRDA (National Research Council, 2013). With the goal of valuing injuries and restoration to improve decision making, the NRC recommended that in addition to documenting injuries to the ecosystem, the trustees should develop (and implement) tools capable of establishing and quantifying the changes in human uses resulting from ecosystem injuries and proposed restoration and also capable of determining the value of the changes to individual communities and society at large (National Research Council, 2013).

The rise of dedicated ecosystem services modeling tools, which represent an alternative to HEA or stated preference methods for valuing ecological losses, is a recent development. This approach is based on production function models that link ecological processes and functions to ecological services, which in turn are linked to indirect and off-site human uses, which in turn are linked to values for the human uses (Bagstad et al., 2013; Guerry et al., 2015). The NRC acknowledged that "the ecosystem approach is still very early in its development and faces many challenges to its implementation, the most serious of which is the lack of comprehensive ecosystem models" (National Research Council, 2013, p. 17).

The tools are currently being used to conduct *ex ante* benefit-cost analysis of policies and projects in federal policies and programs and to inform corporate sustainability decision making. For example, the Bureau of Land Management (part of the U.S. DOI) has evaluated the use of 17 modeling tools, which have been developed by commercial, government and non-profit organizations, to carry out its mandate to manage 99 million ha. of land and 283 million ha. of subsurface mineral rights (Schaefer et al., 2015).

In contrast, ecological services modeling tools have not been widely adopted in the NRDA context to date. An important reason for the difference between administrative versus judicial use is that the standards for admissibility of models (and other "novel scientific evidence") in the courtroom is higher than for their use in agency regulatory decision making (Environmental Law Institute, 2016; Jones, 1997). For admissibility in a courtroom, a judge must make a determination of the relevancy of the evidence and the reliability of the method and its application to the facts. In contrast, in reviewing agency decision making on issues for which they have expertise, the courts defer to the agencies. At this time, the level of uncertainty in the estimates is perceived to be high relative to the admissibility standard, particularly for such a significant case as *Deepwater Horizon*.

And more generally, a second reason for low adoption in NRDA cases is that, for individual cases, the value added by the quantitative tools over the current approach (relying on qualitative assessment of the quality and value of ecosystem services) relative to the additional cost appears to be limited, echoing a conclusion reached for some business applications as well (Waage et al., 2012). First, the geographical scale of analysis is typically quite localized for an NRDA case—both for demonstrating the impacts of the injury and for siting restoration projects that provide the same type, quality, and value of services for the benefit of the humans affected by the injuries.[19] Yet most of the models are not implemented at a sufficiently fine scale to assess site-specific environmental impacts for an individual case. Literature studies, data, and models with qualitative indicators of associated services can be used to provide valuable guidance at lower cost than with a formal ecosystem service model. Second, collecting the data to implement the models is time-intensive, and even more so if new algorithms are required, for example to model the impacts of the specific contaminants or to tailor relationships to the site. And adding a new tier of assessment and conclusions associated with socioeconomic considerations further compounds the uncertainties, while potentially giving a false appearance of precision. As the NRC acknowledges, adding these layers of analysis for an NRDA would significantly drive up costs and extend the time frame for achieving restoration.

Achieving a better understanding of the supply and values of certain ecological services provided by habitats and resources is a desirable goal: such an understanding could change the ways that the public and agencies design and scale projects to restore natural resources. However, this endeavor appears better suited for research conducted outside of the formal constructs of an NRDA and funded by sources other than the responsible parties in the current case (National Research Council, 2013). A critical issue is the lack of predictability in determining where incidents will occur, which impedes targeting of such research. Exceptions may occur in protected areas that are subject to repeated injuries. For example, coral reefs and sea grass beds in several national marine sanctuaries have a history of frequent injuries, each of relatively small scale, but with substantial cumulative impact. NOAA's retrospective studies of recovery rates at injury sites with active restoration funded by NRDA claims, relative to those with natural recovery, have provided the basis for parameterizing HEA models for a continuing stream of cases. With adequate funding, it would be possible to develop extended ecosystem models, including linkages from biological functions to human use and values, that could be used in future cases.

### *5.3. Can HEA be useful in other countries?*

At this time, HEA is the dominant method to quantify damages for ecological losses in the U.S. But does it have relevance elsewhere?

Reflecting a strong U.S. influence, the EU has incorporated resource compensation in its core directive on environmental liability, and EU countries have applied HEA in a number of cases under the directive (Shaw and Wlodarz, 2013). The EU Environmental Liability Directive (ELD) (Council Directive 2004/35/EC), adopted in 2004 and integrated into member states' legislation by 2010, articulates a preference for resource compensation—the costs of remediation measures (the term the EU uses instead of " restoration")—rather than compensation based on the monetary value of the impaired natural resources (ELD art. 8(4)). The ELD imposes liability for the prevention and remediation of certain classes of environmental harm, including damage to species and natural habitats protected under prior directives and, at the option of member states, nationally protected biodiversity, water, and soil/land, including any contamination of land that creates a significant risk to human health.[20] The EU has created guidance materials for developing damage claims using resource compensation approaches, which draw extensively on U.S. methods (Chapman and LeJeune, 2007; Cox, 2007; Edward Brans, 2006).

In contrast, many tropical countries, which are at the center of global biodiversity challenges, include strong natural resource liability provisions in their laws but typically experience challenges in valuing damage claims. A recent review of seven tropical countries (Brazil, Mexico, Democratic Republic of Congo, Nigeria, India, Philippines, and Indonesia)—which span a broad range of legal systems, geography, environmental governance, and human development status—found that all seven maintain statutory regimes creating extensive environmental protections with civil and criminal sanctions (Jones et al., 2015). Six of the countries (all save Nigeria) have enacted additional statutory provisions creating liability for harms to public natural resources. Relative to the U.S. liability statutes, the six countries have generally established liability for a broader scope of environmental harm and have incorporated expanded standing and procedural reforms that increase access to courts.

These statutes consistently focus on restoration or replacement of injured resources, but some do not identify interim lost value as part of damages, or they allow "compensation for damages" when restoration is not possible (Jones et al., 2015). The concept of ecosystem services is frequently introduced, particularly in the more recent statutes, to inform both the restoration of injured resources and the compensation for interim losses, where applicable. Several of the countries, including Indonesia and Brazil, have codified simplified valuation approaches that are essentially lookup tables for literature values per (injured) hectare for different classes of resources and/or ecosystem services drawing inspiration from the initial 1997 publication with estimates of the global value of ecosystems and their services (Costanza et al., 1997). (See Jones et al., 2015, for more details.)

A focus on restoration-based compensation, and the use of HEA to scale compensation projects for injuries to ecological services, could provide these countries with a valuable way forward in addressing their valuation challenges. To move in that direction, we offer several considerations identified in a recent review of liability in a selection of tropical countries (Jones et al., 2015). The first step is to ensure that the statutory measure of damages covers full compensation, for which two dimensions are relevant. This means damages should include compensation for interim losses, as well as restoration of injured resources, or when the injured (or destroyed) resources cannot be restored, replacement or acquisition of the equivalent of the injured (or destroyed) resources. And also damages should cover total losses from resource and ecosystem services losses, including passive uses as well as direct human uses. The second step ensures that the legal provisions create a statutory obligation to spend the monies on restoration or replacement of equivalent resources and establish a trust fund to hold the funds in reserve. The third step is to develop or improve protocols for the assessment of injuries and development of restoration options. Creating staff teams dedicated to natural resource damage claims provides a strong base for building capacity; coordination

[19] Damage assessment for Deepwater Horizon, the largest oil spill in U.S. history, was a challenge of a whole other order of magnitude. Its estimated 134 million gallons of oil (~12 times the size of Exxon Valdez) oiled more than 1300 miles (2092 km) of shoreline, with oil slicks in the Gulf observed cumulatively over time across 43,000 sq. miles (112,000 sq. km) (Deepwater Horizon Natural Resource Trustees, 2016).

[20] ELD compensation includes interim losses in situations concerning bodies of water, protected species, and covered habitats (though not for soil pollution and land damages) (ELD arts. 2(11) and (13), Annex II, para. 1(c) and (d)). In addition, operators can be held liable for the costs of assessing environmental damage; the administrative, legal, and enforcement costs; the costs of data collection and monitoring; and oversight costs (ELD arts. 2(16), 8(1)). The other cited EU directives include Council Directive 92/43/EEC 1992 (discussing the conservation of natural habitats and wild fauna and flora), Council Directive 2000/60/EC, 2000 (establishing a framework for community action in the field of water policy), and Council Directive 2009/147/EC 2010 (discussing the conservation of wild birds). *See* (Edward Brans, 2006).

with other national or regional initiatives that conduct resource inventories and/or develop region or resource-based restoration plans is critical. The guidance documents developed in the U.S. and EU can provide a foundation.

## 6. Conclusion

The U.S. restoration-based damage measure, first codified in the OPA regulations (1996), is a widely accepted legal innovation capable of producing more expeditious case settlements and more timely restoration. First, it focuses injury determination—from the beginning of the case analysis—on informing the design of restoration. Second, it provides a framework for valuing ecological services that deflect controversy from the use of stated preference methods; instead, the measure of damages is the cost of primary restoration to restore or replace the injured or destroyed resources, plus the cost of compensatory restoration to compensate for interim losses.

For injuries to ecological services that provide indirect or passive human uses—which are not easily quantified and are the focus of this paper—trustees have relied almost exclusively on habitat equivalency analysis (HEA), a simplified "service-to-service" approach. Analogous to in-kind trading, which does not require valuation, this approach was designed for compensatory restoration projects that provide resources and ecological services of the *same type and quality* and *comparable value* as those compromised or destroyed. Natural resource trustees initially applied HEA to vessel groundings or oil spills with limited injuries. Over the past 25 years, its use has expanded to include larger and more complex cases, including ones with multiple toxic contaminants and impairments to multiple habitats and resources. Furthermore, HEA has been applied when it was not feasible or cost-effective to provide restoration projects supplying the *same types* of resources because (1) the contamination area was so vast and the injury so entrenched or (2) the injured habitat could not be restored (e.g., nearshore sediments) or was of such marginal value that the ecosystem services could be replaced more effectively with another habitat of higher value, such as wetlands. In these cases, the trustees focus on replacing the same or comparable types of *services*.

Two factors are key to using HEA: the choice of restoration project and the choice of metric for measuring ecological services. The selection of restoration projects and sites is critical to achieving comparability in the *quality and value of services* at the injury and restoration sites. Trustees must consider both whether the restoration project site has comparable capacity, opportunity, and payoff for ecological services and whether the distribution of the benefits is equitable. The second factor, the choice of metric, must capture the difference in quality and service characteristics between the injury and restoration sites. The scale of restoration projects will also be influenced by the choice of other HEA parameters, including recovery functions and interest rates.

When the regulations were written, it was envisioned that stated preference methods would be a valuable tool to estimate the cross-resource or cross-ecosystem service tradeoffs needed for compensatory projects that differ in type and quality from the injuries. In practice, however, the use of stated preference methods has been limited in part because they are more costly and more controversial than the simplified (service-to-service) approaches. Another major limitation is that stated preference methods are constrained in the number of tradeoff parameters that can be estimated because of perceived respondent burden in conducting surveys. As a result, it is difficult to glean insights about the value of design tradeoffs at the level of detail needed to inform restoration planning. Public participation in restoration planning is an alternate source of guidance regarding public preferences.

Another potentially useful method for scaling compensatory restoration, which has undergone dramatic advances since the OPA regulations were finalized in 1996, is production-function-based ecosystem services modeling. Ecosystem services modeling holds great promise in the future to capture greater detail about ecosystem functions and services than HEA currently captures (or than stated preference surveys could capture). However, at the current time the potential added value in individual legal cases has been judged to be limited relative to the method's costs, given the current state of uncertainty in the modeling.

The restoration-based measure of damages and the HEA methodology to implement it are now widely used for cases brought under all NRDA statutes in the U.S. to develop compensation claims for injuries to ecological services. This approach was adopted by the EU's ELD of 2004, and it may provide a useful framework for other countries.

## Disclaimer

The views expressed are the authors' and should not be attributed to NOAA or ELI.

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## Appendix A. Habitat equivalency analysis case study: Blackbird Mine hazardous material release

**Summary of legal actions:** For remedies to hazardous contamination brought under CERCLA, U.S. EPA typically issues a record of decision (ROD) for the responsible party to perform remedial actions to clean up the contamination. In addition, as needed—depending upon the state of impaired resources following the remedial action—the natural resource trustees will bring a claim for natural resource damages to restore injured or destroyed resources and compensate for interim losses.

In this case, an NRDA consent decree was entered into federal district court in 1995, while the remedial actions (managed by U. S. EPA and the state of Idaho) were still in the planning stage. In the consent decree, the settling defendants committed to (1) restore water quality in Panther Creek and Big Deer Creek in Idaho to a level that will support all life stages of salmonids, (2) implement a restoration plan for returning Snake River Chinook salmon to Panther Creek and (3) implement future remedial actions under a separate order or consent decree. Full implementation of the restoration plan was conditional upon completion of the remedial action.

U.S. EPA issued the ROD for remedial actions in 2003 and issued administrative orders for the Blackbird Mine Site Group to implement remedial designs and action consistent with the ROD at several points between 2003 and 2013. A first round of remedial actions was completed by August 2012; questions remain as to whether the water quality has returned to target levels. The natural resource trustee restoration plan has been initiated, and salmon have returned to the Panther Creek; questions remain about the genetic diversity of the returning populations, which can interfere with the adaptability of the salmon to the ecosystem (Blackbird Mine Natural Resource Trustees, 1995; CH2MHILL, 2013; Chapman et al., 1998; US EPA Office of Environmental Cleanup, 2003).

### A.1. Site characteristics and source of contamination

Located in the Panther Creek drainage basin of the upper Salmon River, the Blackbird Mine site in east central Idaho includes underground tunnels, an 11-acre open pit, millions of tons of tailings and more than 2 million cubic yards of waste rock (Fig. A1). Panther Creek historically was the fourth most important source of Chinook salmon among the Salmon River tributaries. Toxic releases from the copper mine have contributed to the decline of resident trout species and the ultimate elimination of threatened spring and summer Chinook salmon and steelhead from Panther Creek. Surface water and groundwater continue to flow through the mining tunnels, discharging contaminants into the watershed. In addition, the operation of Blackbird Mine resulted in further fishery habitat degradation and loss through the construction of access roads, timber clearing, sedimentation and creek channel re-alignments.

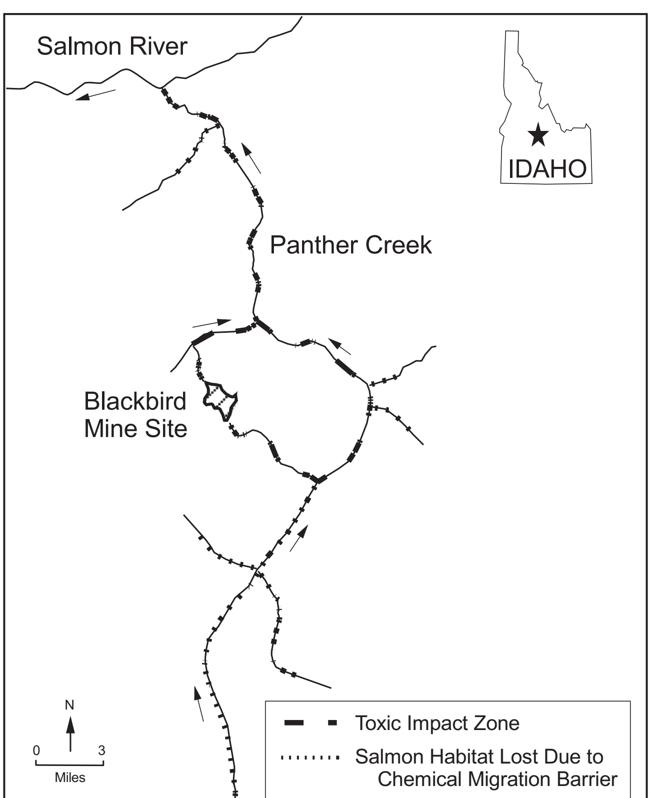


**Fig. A1.** Blackbird Mine site in Panther Creek: lost salmon habitat (Chapman et al., 1998).

### A.2. Resource injuries and lost services

Panther Creek is one of the more accessible streams in Lemhi County, Idaho, and once provided recreational opportunities including fishing, birding and picnicking. Moreover, Panther Creek supported subsistence fishing for Native American tribes, which depended on the fish resource as a food, barter, and religious base for their culture. Since the 1960s, the annual number of salmon has declined gradually throughout the upper Salmon River basin, reflecting the effects of dam construction and habitat change on the Columbia and Snake Rivers. However, mining caused the loss of the Panther Creek salmon *before* the impacts of downstream dams; despite the closure of Panther Creek for salmon fishing in 1957 in an effort to preserve the remaining run, the spring and summer Chinook salmon and steelhead populations plummeted to zero in 1963, with only sporadic sightings since that time. Salmon fishing has been closed throughout Idaho since 1977.

The natural resource trustees chose three primary indicators of the ecosystem health to focus on in the injury quantification: exceedance of surface water criteria; injury to streambed food web species; and injury to resident and anadromous fish. Impacts to the aquatic ecosystem not related to Blackbird Mine were excluded from the injury determination. Trustee scientific studies concluded that mining operations had reduced fish populations and their prey and had contaminated sediments and prey species with toxic concentrations of metals over a 25-mile distance from the mine to the Salmon River.

### A.3. Damage claim: costs of the restoration plan

#### A.3.1. Costs of restoring injured resources to baseline

In the innovative trustee restoration plan, the potentially responsible parties agreed to incorporate achievement of water quality goals (using an unspecified remedial action) as the foundation to restoration activities. The restoration plan focused on returning approximately 200 spawning adult Chinook salmon each year, based on scientific studies that estimated this is the capacity of Panther Creek, given the other factors that have reduced spawning populations throughout the Columbia River Basin, in which it is located. It was anticipated that other injured species would quickly re-populate Panther Creek after the water quality is restored. However, Chinook salmon return only to their natal streams to spawn. Consequently, with the original Panther Creek stock extirpated, restoring the salmon run was envisioned to require the use of a hatchery to culture select wild donor stocks that can be introduced into Panther Creek. In addition, primary restoration included some streamside stabilization and restoration work to increase the survival rate of Chinook young.

#### A.3.2. Costs of compensatory restoration

Trustees used HEA to calculate the amount of additional restoration work that would be sufficient to compensate the public for the interim losses. The compensatory projects, located both within and outside of the Panther Creek drainage basin, include channel meander reconstruction, fencing to protect sensitive riparian habitats and construction of off-channel rearing ponds. These measures were selected as the most beneficial and cost-effective for enhancing populations of salmon, trout, steelhead, and other desirable attributes of the Panther Creek system.

### A.4. Current status of EPA remedial actions and trustee restoration actions

As a result of the extensive remedial action and trustee restoration efforts, habitat conditions have improved dramatically in Panther Creek to a level suitable for natural production of Chinook salmon and steelhead. However, the remediation has not been completely successful in achieving its water quality goals; further remedial actions are being considered.

The compensatory restoration projects to improve spawning and rearing habitat are operational. For example, physical and biological monitoring data show that hydric vegetation has increased at the majority of the monitoring sites and bare soil has decreased at some of the sites as a result of the cattle exclusion project (Blackbird Mine Natural Resource Trustee Council, 2012).

Recent research has found that anadromous Chinook and steelhead have re-occupied Panther Creek in densities, biomass, year class strength and condition factors that are similar to reference sites without mining influences (Mebane et al., 2015). As anticipated, fish stocks from nearby creeks have not strayed into Panther Creek; the returning chinook salmon are from three different hatchery populations stocked by Idaho Fish and Game starting in 2001 with some apparent strays from the South Fork Salmon River major population group. However, there are potential issues with genetic diversity, which can interfere with the adaptability of the salmon to the ecosystem. The Shoshone Bannock Tribes have an active sampling and genetic texting program to complement their stocking in the area (Smith et al., 2012).

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