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Special Series

Translating Ecological Risk to Ecosystem Service Loss

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EDITOR'S NOTE:

This is 1 of 4 papers reporting on the results of a SETAC technical workshop titled "The Nexus between Ecological Risk Assessment and Natural Resource Damage Assessment under CERCLA: Understanding and Improving the Common Scientific Underpinnings," held 18–22 August 2008 in Montana, USA, to examine approaches to ecological risk assessment and natural resource damage assessment in US contaminated site cleanup legislation known as the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA).

ABSTRACT

Hazardous site management in the United States includes remediation of contaminated environmental media and restoration of injured natural resources. Site remediation decisions are informed by ecological risk assessment (ERA), whereas restoration and compensation decisions are informed by the natural resource damage assessment (NRDA) process. Despite similarities in many of their data needs and the advantages of more closely linking their analyses, ERA and NRDA have been conducted largely independently of one another. This is the 4th in a series of papers reporting the results of a recent workshop that explored how ERA and NRDA data needs and assessment processes could be more closely linked. Our objective is to evaluate the technical underpinnings of recent methods used to translate natural resource injuries into ecological service losses and to propose ways to enhance the usefulness of data obtained in ERAs to the NRDA process. Three aspects are addressed: 1) improving the linkage among ERA assessment endpoints and ecological services evaluated in the NRDA process, 2) enhancing ERA data collection and interpretation approaches to improve translation of ERA measurements in damage assessments, and 3) highlighting methods that can be used to aggregate service losses across contaminants and across natural resources. We propose that ERA and NRDA both would benefit by focusing ecological assessment endpoints on the ecosystem services that correspond most directly to restoration and damage compensation decisions, and we encourage development of generic ecosystem service assessment endpoints for application in hazardous site investigations. To facilitate their use in NRDA, ERA measurements should focus on natural resource species that affect the flow of ecosystem services most directly, should encompass levels of biological organization above organisms, and should be made with the use of experimental designs that support description of responses to contaminants as continuous (as opposed to discrete) variables. Application of a data quality objective process, involving input from ERA and NRDA practitioners and site decision makers alike, can facilitate identification of data collection and analysis approaches that will benefit both assessment processes. Because of their demonstrated relationships to a number of important ecosystem services, we recommend that measures of biodiversity be targeted as key measurement endpoints in ERA to support the translation between risk and service losses. Building from case studies of recent successes, suggestions are offered for aggregating service losses at sites involving combinations of chemicals and multiple natural resource groups. Recognizing that ERA and NRDA are conducted for different purposes, we conclude that their values to environmental decision making can be enhanced by more closely linking their data collection and analysis activities.

Keywords: Ecological risk assessment Natural resource damage assessment Ecosystem services Assessment endpoints Environmental decision making

INTRODUCTION

Environmental management of hazardous sites in the United States often involves 2 activities legislated in the Comprehensive Environmental Response, Compensation,

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and Liability Act (CERCLA, or Superfund): remediation of contaminated environmental media, and restoration of or compensation for injured natural resources and losses of the services they provide. Remediation is usually overseen at the federal level by the US Environmental Protection Agency (USEPA); restoration of injured natural resources is conducted by certain federal, state, and tribal groups (Natural Resource Trustees) through the natural resource damage assessment (NRDA) process. Superfund remedial decisions are informed by the remedial investigation/feasibility study (RI/FS) process which includes ecological risk assessment (ERA). The focus of the ERA is to determine the risk to ecological receptors posed by chemical and physical stressors at a Superfund site and ultimately to inform clean-up decisions. In NRDA, Trustees quantify the magnitude of injury (impact) sustained by natural resources and the services they provide due to the release of oil or hazardous substances and scale the damage claim to provide the appropriate amount of restoration. The technical information required by the 2 assessment processes is similar in many regards and includes the distribution and concentrations of the chemicals of concern, the actual or potential degree of exposure by ecological receptors, and the potential for or measurement of adverse effects resulting from these exposures (see Gala et al. 2009).

The CERCLA provides, and in certain cases requires, that an assessment of natural resource damages be brought after the remedy selection process, reasoning that until a remedy is selected by the USEPA, the Trustees lack sufficient information to reasonably evaluate and seek appropriate compensation for injured natural resources and the services they provide. Until recently, the ERA and NRDA processes have been conducted independently of one another at sites where both are conducted, with the ERA often being completed before the NRDA begins. While acknowledging that Superfund ERA and NRDA have distinct purposes under CERCLA (see also Gala et al. 2009; Gouguet et al. 2009), it is reasonable to evaluate the extent to which the 2 processes can and should be mutually supportive. It is becoming clear to practitioners of ERA and NRDA that there is considerable overlap in the types of data needed to inform decision making. Although a limited portion of the data collected and analyzed for an ERA has been used to inform the NRDA process, for the most part ERA information has not been of sufficient specificity or robustness to address most of the needs of the NRDA. Some of the common elements of ERA and NRDA, such as collecting and assessing environmental data, have been described (Barnthouse and Stahl 2002). Yet, neither the scientific underpinnings of these overlaps nor the degree or magnitude of distinctions between the data needs of the 2 processes have been evaluated critically.

A SETAC technical workshop was convened to discuss how ERA and NRDA data needs and assessment processes could be more closely linked (Stahl et al. 2009). The attendees of the workshop included ERA and NRDA practitioners from the public and private sector, many of whom have been on opposite sides of contentious, even litigious, NRD cases. The workshop built upon previous efforts to enhance coordination of environmental response and natural resource restoration, including the US Department of the Interior (DOI) Natural Resource Damage Assessment and Restoration Advisory Committee and the ongoing Trustee/Industry sponsored west- and east-coast Joint Assessment Team meetings. Overall, the workshop was intended to advance the dialog among practitioners about how the processes of ERA and NRDA at hazardous sites might be coordinated better, with a goal of enhancing the efficiencies and effectiveness of analyses supporting environmental decisions.

This is the 4th paper from this workshop. Our objectives with this paper are to evaluate the technical underpinnings of recent methods used to translate measurements of ecological risk and natural resource injury into ecological service losses and to propose ways to enhance the usefulness of data obtained in ERAs to the NRDA process. Three aspects are addressed: 1) improving the linkage among ERA assessment endpoints and ecological services evaluated in the NRDA process, 2) enhancing ERA data collection and interpretation approaches to improve translation of ERA measurements in damage assessments, and 3) highlighting methods that can be used to aggregate service losses across contaminants and across natural resources. We propose that ERA and NRDA both would benefit by focusing on ecological assessment endpoints that correspond most directly to those considered in restoration and damage compensation decisions. A thesis central to this paper is that ecosystem services can be a common currency used by both processes to guide environmental decision making.

We begin by describing the concept of ecosystem services and how their adoption as assessment endpoints in ERAs will facilitate quantification of service loss in the NRDA process. Here, we build upon the USEPA's (2003) efforts to promote generic ecological assessment endpoints for ERA by suggesting development of generic ecosystem service assessment endpoints that we believe will both enhance the value of ERA to risk management and enhance the value of these data to a NRDA. After illustrating how information about risk to ecosystem service assessment endpoints can be used in NRDA, we turn our attention to issues surrounding the translation of measurement endpoints to service losses for NRDA. Consideration is given to some of the qualities desirable in ERA measurement endpoints that facilitate quantifying service loss, as well as to some of the important issues affecting the translation. Biodiversity is then offered as a key measurement endpoint, showing considerable promise for linking measurable effects to ecosystem service risk and service loss. We next consider the problem of aggregating service losses across contaminants and natural resources, offering recent case studies and approaches for aggregation. We conclude with key observations and recommendations from our workshop discussions for enhancing the use of Superfund ERA data in a NRDA.

ECOSYSTEM SERVICES AS A COMMON CURRENCY

There is growing awareness within the ecological and social science communities that improved environmental management can be achieved by considering environmental systems holistically (e.g., Di Giulio and Benson 2002). This view holds that functioning ecosystems contribute to the well-being of ecological and social components of the larger environmental system and considers humans to be an explicit part of that system (Miranda et al. 2002). Reflected in this systems perspective is the concept that the structural components and processes of intact ecosystems interact functionally to provide the support required by all life within the system. In a broad sense, the contributions of ecological systems to the vitality of human and nonhuman species alike can be considered ecosystem services, that is, the benefits received from properly functioning ecosystems that contribute to the well-being of living organisms. Generally included in this definition is the provisioning of goods, such as food, fiber, shelter, and clean water, and the processes regulating biological productivity, material cycling, climate, and so on. Recent attention has been given to the paramount importance of ecosystem services to humans and society (e.g., Daily 1997; Daily et al. 1997; Millennium Ecosystem Assessment 2005; Stahl et al. 2007). Although the perspective taken in these discourses is decidedly anthropocentric, the notion that ecological functions produce the ecosystem services from which humans benefit reflects the implicit consideration that nonhuman species also derive benefit from functioning ecosystems. Managing the environment with this systems view in mind is likely to yield greater rewards to humans and other organisms than does a reductionist approach that focuses on individual ecological receptors or particular structural components in isolation from the larger environmental system.

The goals of the NRDA process are to return natural resources injured because of the release of hazardous substances to their uninjured or baseline condition (the condition but for the release of hazardous substances) through direct restoration or replacement of injured resources and to compensate the public for service losses occurring until those injured resources are restored. Ecological injuries are quantified in terms of the reduction in the physical, chemical, or biological services the natural resources provide, and compensation for those injuries are claimed in terms of damages (monetary) or directly as restoration actions. Damages are calculated with the use of various market and nonmarket economic techniques, and both damages and direct restoration projects are scaled to the magnitude of the injury claim.

The objectives for ERA conducted under CERCLA and similar state statutes are to identify and characterize the current and potential threats to the environment from a hazardous substance release and to identify cleanup levels that would protect those natural resources from additional adverse effects (USEPA 1997a). Although the intention of Superfund ERA is to provide information about contamination risk to societally relevant assessment endpoints (e.g., the abundance of small mammal populations), the relationships among ERA assessment endpoints and valued ecosystem services often go unstated in practice. Furthermore, insufficient attention has been given to the relationships between measurement endpoints (termed measures of exposure and effect in USEPA 1998) and ecosystem services to facilitate straightforward translation of adverse ecological effects to service losses in NRDA. Recognition and selection of ecological assessment endpoints that explicitly and more directly reflect ecosystem services should improve the value of ERA data to the NRDA process and likely will improve the societal relevance of ERA conclusions to remediation decisions.

Ecosystem service assessment endpoints

The USEPA (1998) defines an assessment endpoint to be an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes. The meaning of the term ecological entity in this definition is intentionally broad, to include a species, a specific habitat, or an ecological function. For any particular site-specific ERA, assessment endpoints are identified specific to the ecology of the site and the chemical stressors present (USEPA 1997a). An example assessment endpoint for an aquatic site ERA might be benthic invertebrate community (the entity) diversity (its attribute). In this case, the ERA might consider the effects of sediment contamination on benthic invertebrate community diversity by evaluating various measurement endpoints, such as the number of species counted in benthic samples, as a function of chemical concentration in the sediment. Information about risk to this assessment endpoint would be used by site decision makers as they consider the need for sediment remediation and the options for cleanup. Ecological relevance, susceptibility to the stressor, and relevance to management goals are the key considerations when selecting assessment endpoints responsive to the needs of the decision maker (USEPA 1998). Attention to the 1st 2 of these helps to ensure the scientific credibility of the ERA; attention to the 3rd enhances the significance of assessment results to decision makers and the public.

Explicitly linking ERA assessment endpoints to data needs of the damage assessment process will enhance the likelihood that ERA data collection and analysis activities will provide information useful for both remediation and restoration decisions. The USEPA's (1997a, p. I-4) guidance for Superfund ERAs identifies valued ecological resources (i.e., entities of assessment endpoints) to include "those without which ecosystem function would be significantly impaired, those providing critical resources (e.g., habitat, fisheries), and those perceived as valuable by humans" in its interpretation of assessment endpoint, any of which should contribute to injury quantification and damage determination. In practice, the ecological entities identified in Superfund ERA assessment endpoints tend to be structural components of ecosystems. Consider, for example, the 8 assessment endpoints selected for the 2000 revised baseline ERA for the Hudson River polychlorinated biphenyls (PCBs) site in New York State (http://www.epa.gov/hudson/revisedbera-text.pdf):

- Sustainability of a benthic community structure, which is a food source for local fish and wildlife;
- Sustainability (i.e., survival, growth, and reproduction) of local fish (forage, omnivorous, and piscivorous) populations;
- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous bird populations;
- Sustainability (i.e., survival, growth, and reproduction) of local waterfowl populations;
- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous bird populations;
- Sustainability (i.e., survival, growth, and reproduction) of local insectivorous mammal populations;
- Sustainability (i.e., survival, growth, and reproduction) of local omnivorous mammal populations; and
- Sustainability (i.e., survival, growth, and reproduction) of local piscivorous and semipiscivorous mammal populations.

Although reflecting the interests and values of the parties involved, all of these assessment endpoints describe structural components of the Hudson River ecosystem, and only the 1st (benthic community structure) is linked explicitly in its statement to an ecosystem service (food source for fish and wildlife) as defined here. Data describing impacts to structural components of ecosystems can be used to identify and quantify natural resource injury and determine monetary damages or restoration requirements. Assessing damages can be fairly straightforward when those components are traded as goods in markets (albeit not without philosophical controversy; see White 1990), such as with some of the fish species included in the 2nd assessment endpoint above. However, quantifying damages in monetary terms is not so straightforward when no markets exist for the injured resource or the ecosystem services they provide. Developing and scaling a damage claim or restoration project on the basis of information from ERAs that are only loosely or indirectly linked to ecosystem services becomes particularly problematic.

Translations between ERA and NRDA could be more straightforward if ERA assessment endpoints were couched explicitly in terms of ecosystem services. With ecosystem services specifically in mind, generic ecological assessment endpoints (GEAEs) could be identified that can be tailored to the decision support needs of individual sites. The GEAEs are broadly described assessment endpoints (e.g., abundance of an assessment population) that can be applicable in a variety of environmental management contexts (USEPA 2003; Suter et al. 2004). The USEPA (2003) developed an initial set of 15 GEAEs to help guide planning of the ecological risk assessments that support the array of the Agency's environmental protection decisions, including those of Superfund. The GEAEs in this set were selected after consideration of their usefulness in informing USEPA decisions, the practicality of their measurement, and the clarity with which they can be defined. Several, if not all, of the GEAEs in this initial set already appear to be responsive to the needs of NRDA (also see Gala et al. 2009). Importantly, USEPA (2003) describes the relationships between the individual GEAEs and several of the environmental values that the public ascribes to ecological entities and functions. Having this relationship well described is particularly relevant to the translation questions of concern here, because linking assessment endpoints to public values can help identify economic methods appropriate for monetary damage determinations.

The USEPA (2003) encourages development of additional GEAEs to enhance their coverage of assessment scenarios. We recommend that NRDA and ERA practitioners jointly review the initial set of GEAEs to identify those best suited to serve as generic ecosystem service assessment endpoints and to specify other GEAEs that could enhance translation of ecological risk estimates to ecosystem service losses. Nationally, a team of ecological risk assessors, Trustees, regional Biological Technical Assistance Group (BTAG) members, and other stakeholders could be convened to expand the list of GEAEs to include those particularly responsive to the needs of NRDA at local and national scales. An explicit focus on ecosystem service assessment endpoints (ESAEs) in ERA is consistent with the increasing emphasis on the role of ecosystem services in environmental management decisions (e.g., Daily et al. 1997; Millennium Ecosystem Assessment 2005; Dale et al. 2008) and thus should increase the value of assessment results to decision making more generally. A similar recommendation was made by USEPA (2006) for development of generic endpoints that encompass key ecosystem goods and services for routine consideration in the ecological benefits assessments that support benefit–cost analyses of environmental policy and regulation.

Using ecosystem service assessment endpoints to inform NRDA

Two approaches that are employed in NRDA illustrate how ERAs that focus on ESAEs can inform restoration and compensation decisions directly. The 1st, habitat equivalency analysis (HEA), is a method used to determine the appropriate type and scale of compensation for loss of natural resource habitats and the ecological services they provide (see Unsworth and Bishop 1994; Dunford et al. 2004; NOAA 2006). The principal concept underlying HEA is that the public can be compensated for service losses through habitat replacement or restoration projects that provide resources and services of the same or appropriately similar type as were lost. The HEA addresses differences in the types and levels of services provided by a habitat before injury and after restoration and provides a framework for scaling restoration projects to account for any differences. Scaling considerations include interim service losses or gains that occur in the time interval between injury and recovery to the baseline conditions. In practice, future service gains and losses are discounted on the basis of the economic theory that consumers prefer to use their commodities, in this case the services provided by natural resources, in the present rather than at some time in the future. Thus, future service gains or losses are discounted to reflect their present worth, and the results of HEAs are discussed in terms of discounted service acre years (DSAYs), discounted stream mile years, and so on (see NOAA 1999 for a complete description of discounting).

The ERAs that are focused on the ecosystem services provided by functioning habitats can have greater value to HEA-based NRDAs than do those focused on more traditional assessment endpoints. An ESAE described in the form of a habitat type, say a wetland (the entity), and services such as biological productivity or nutrient retention (the attributes), is likely to concentrate ERA analysis activities on providing the data needed to characterize risk to habitat services in a manner that is more integrated than would be assessment endpoints that describe structural elements of the system in isolation. Because the needs of the 2 assessments are similar, ERA measurements of adverse impacts to those services would inform quantification of service flow reductions in a HEA-based NRDA directly. The comparability of endpoints of the 2 assessment processes, together with the more integrated nature of ESAEs, can help overcome some of the difficulties associated with aggregation described later in this paper.

The 2nd NRDA approach discussed here, resource equivalency analysis (REA), is used to scale the injury when it primarily involves 1 or more natural resource species rather than a habitat. In REA, the injury typically is measured in terms of number of individuals killed or loss of reproductive capacity. Data on life history characteristics of the resource, such as survival rate, average life expectancy, average reproductive rates, age of injured/killed individuals, etc., are used to estimate the total impact of the injury. Like HEA, the economic model behind REA calculates the present value of the injured resource (service flow losses) and the restored resource (service flow gains). Instead of calculating DSAYs of injury, however, the REA model calculates lost organism years, which is an integration of the injury to the resource over time on the basis of basic life history characteristics. The REA estimates the difference between 2 population trajectories for the injured species: the trajectory for the population that would have occurred without the injury and one that estimates the population trajectory for the injured population. The difference between these 2 trajectories often is represented as discounted lost organism years. Restoration projects can then be scaled to provide an equivalent replacement of the estimated 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. The HEA can be used to scale injuries to a single biotic resource by translating those injuries to a habitat loss, but REA provides a more direct measurement of loss when individual species are injured. Additional discussion on the development and application of REA in the NRDA context can be found in Donlan et al. (2003), McCay et al. (2004), and the damage assessment and restoration plan for the Luckenbach incident (CADFG 2006).

The resource species on which REA focuses can be thought of as goods from the ecosystem services perspective. And in many regards, ESAEs describing the attributes of such goods would be similar to traditional assessment endpoints like most of those selected for the Hudson River PCBs site listed above. With ESAEs articulated at appropriate levels of biological organization (specifically, population and community levels), ERA analysis activities would be better positioned to provide the information needed directly by NRDA to quantify injury and service flow reductions associated with resource species. When viewed from a community perspective, the adverse impacts to resource species measured in ERA can be considered in NRDA in the context of other resource species and their functional relationships. Furthermore, the data and models developed for the ERA could be directly applicable to calculation of service flow gains over the time period of natural resource restoration and recovery, if practicable. Additionally, careful framing of the ESAE can add value to observations of bioaccumulation and tissue contamination at sites—notoriously challenging measures to interpret in ERA in terms of adverse impact—because these data can be considered in terms of lost ecosystem goods when contamination of food stocks is known to be deleterious.

TRANSLATING MEASUREMENT ENDPOINTS TO ECOSYSTEM SERVICE LOSSES

In some cases, risks to assessment endpoints cannot be assessed directly in an ERA and must be inferred from changes in measurement endpoints (measures of exposure and effect) used as surrogates. This situation is likely to persist even with a focus on ESAEs. In NRDA, baseline and lost services are often estimated from data obtained in the ERA, but the linkages between measurement endpoints and service endpoints can be tenuous. Thus, both processes will be served by continued attention to improving the translational linkages among measurement endpoints and ecosystem services. In this section, we consider some of the key issues affecting translation of ERA measurement endpoints to ecosystem service losses and how the translations might be improved. We conclude the section by promoting biodiversity as a measurement endpoint with inherent linkages to ESAEs.

Comparability of data needs of ERA and NRDA

Ecological risk assessment and NRDA have different roles in the management of hazardous sites, but in many regards, they have similar informational requirements. Both assessment processes require data describing exposure pathways, environmental exposure concentrations, the toxicity of chemicals and the effects associated with exposure. Although there are limitations to the direct use of some data collected for ERA in NRDA (Gala et al. 2009), much of the raw data generated during ERA likely can prove useful in an injury assessment. For instance, both the ERA and NRDA likely would be informed by data from sediment samples regarding exposure concentrations. Focusing on the commonalities in data requirements between the 2 assessment processes might provide an opportunity for developing an integrated data collection program at hazardous sites.

Despite the commonalities that exist, NRDA has several unique data requirements for which data developed to evaluate risk to assessment endpoints (i.e., measure endpoints) often are insufficient, including the extent of temporal and spatial scale of contamination and the nature of the stressors evaluated. For example, baseline ecological risk assessments focus on estimating risks associated with current conditions at the site, whereas Trustees working on NRDA cases are, by statute, permitted to seek damages for past injuries resulting from a chemical release as well as for future injuries that likely will occur until the injured resources return to baseline conditions. In addition, loss of ecological services associated with remedial activities and the implementation of the remedy is also compensable under NRD statutes. Postremedy monitoring or confirmation data obtained in the remedial process likely will not be adequate to quantify future losses of ecological services resulting from residual contamination or from remedial activities. Furthermore, the focus of information gathering for a hazardous site ERA is most often restricted to a well-defined area, one that is limited by regulatory or policy considerations. Natural Resource Damage Assessment investigations can extend to wherever the site contamination has resulted in natural resource injury, which could be well beyond the boundaries of the Superfund site. For example, investigations of locations with historical levels of contamination that attenuated over time, or of widely ranging receptors such as birds, likely would be omitted from an ERA but might be part of the NRDA data needs. Finally, information usually is not collected in remedial investigations about alterations to habitat and the presence of stressors other than chemicals that might be affecting natural resources or the services they provide. In contrast, in the NRDA process, these types of information often form the basis for establishing baseline conditions at a site.

Characteristics of broadly valuable measurement endpoints

From the standpoint of NRDA, some types of data obtained in Superfund ERAs are more useful than are others. Because much has been said about characteristics of measurement endpoints desirable from the standpoint of their use in ERA (e.g., USEPA 1998), here we consider some key issues relative to applying ERA measurement endpoints to the NRDA process.

Natural resource species as endpoint entities—The relationship between measurement endpoints and ESAEs is inherently uncertain. One of the most important challenges is to identify the natural resource species that are most likely to influence the ecosystem processes and functions that determine ecosystem services. For example, it is highly unlikely that species loss from contaminated ecosystems is a random process. The susceptibility of a species to contaminants and the likelihood of local extinction are influenced by a wide range of species traits, including mobility, longevity, reproductive rates and body size, and species-specific susceptibility to the type and concentration of the contaminant (Raffaelli 2004; Solan et al. 2004; Bunker et al. 2005). Species traits also modify resource dynamics, trophic structure, and disturbance regimes and therefore will influence the relationship between biodiversity and ecosystem processes (Chapin et al. 1997). Raffaelli (2004) provides a conceptual model to show how species traits that determine vulnerability to anthropogenic stressors vary among trophic levels. This model could be used to predict which species are most likely to be eliminated from an ecosystem and the potential cascading effects on ecosystem processes.

Levels of biological organization—The appropriate level of biological organization for assessing effects of contaminants has seen significant discussion in the ecotoxicological literature (e.g., Adams et al. 1992; Clements and Kiffney 1994; Clements and Newman 2002; Barnthouse et al. 2007). In general, the level of biological hierarchy examined is inversely related to the degree of mechanistic understanding of stressor effects and causation. However, responses at higher levels of biological organization are more ecologically meaningful and, when established within the context of NRDA, typically lead to larger damage claims. Because of the increasing complexity and cost of trying to understand and establish cause and effect relationships between a particular contaminant and higher levels of biological organization, most ERAs and NRDAs tend to focus on establishing mechanistic relationships at lower levels. If such a relationship can be established, then further work might be conducted to establish the larger ecological importance of that lower level effect. In Figure 1, we present 3 examples in which responses at lower levels of biological organization are directly or indirectly linked to ecosystem processes. Elevated levels of PCBs in migrating salmon can result in reduced reproductive success, lower population density, and fewer salmon returning to their native streams. Lower numbers of returning salmon can have significant consequences for nitrogen export to adjacent riparian and terrestrial ecosystems (Naiman et al. 2002). Mayflies can be an important seasonal food item for many aquatic and riparian stream species. Reduced genetic diversity of mayflies in a metal-polluted stream increased the susceptibility of these insects to novel stressors (e.g., acidification, ultraviolet-B radiation) and resulted in lower ecosystem resilience (Clements 1999; Courtney and Clements 2000; Kashian et al. 2007). Finally, numerous studies have demonstrated a direct relationship between species diversity and primary production in plant communities (see review by Hooper et al. 2005). Contamination can cause local extirpation of species, thereby reducing local species diversity and primary production. Each of these examples includes a measurement endpoint that would be appropriate for an ERA. Each measurement endpoint is also associated with an important ecosystem process. Although it is true that measurements of higher levels of biological organization (e.g., species diversity and community composition) can be directly linked to ecosystem processes, the above examples show it is also possible to make this link with measurements at lower levels. We suggest that ERA investigations could further benefit NRDAs by carefully selecting the lower level relationships examined in risk determinations. By thoughtfully selecting those species that might provide a closer link to ecosystem services, the ERA would produce a more robust understanding of risk in relation to the needs of remediation decisions and provide data of high value to the NRDA process.

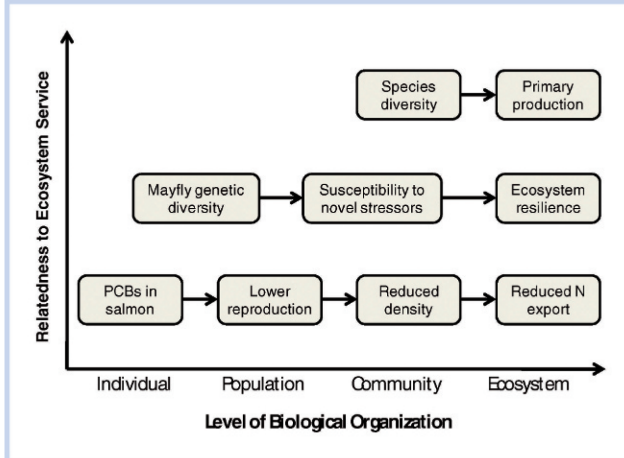

Figure 1. The relationship between ecological risk assessment (ERA) measurement endpoints at different levels of biological organization and ecosystem processes.

Treatment of toxicity data—Studies conducted to evaluate changes in measurement endpoints (e.g., toxicity, behavioral, genomic, or field studies) are often designed to determine a given statistical endpoint or a discrete effects threshold. Test groups of animals are exposed to different concentrations of chemicals, and different effects, such as survival, growth, or reproduction, are monitored. A statistically defined no-observed-effect concentration (NOEC) typically is calculated from these tests. Such statistics are used commonly in screening-level hazard quotient assessments but have limited value for quantitative risk assessment or NRDA purposes. They do not support spatial–temporal evaluation, nor do they provide a means to assess the extent and severity of injury. The use of NOECs has been severely criticized on statistical grounds (Laskowski 1995; Suter 1996; Van der Hoeven et al. 1997), and it has been concluded that the use of NOEC values should be abandoned (OECD 2006). The proposed alternative is to use methods designed to produce continuous data, such that exposure–response and other types of continuous variable data relationships can be developed. This approach can be used for lethal, sublethal, behavioral, genomic, and many other types of toxicity tests. The use of the entire exposure–response range allows calculation of an effect level associated with any given exposure concentration (EC_x). Values of EC_x depend on the exposure time (Jager et al. 2006), and EC_x values decrease asymptotically with increasing exposure time because time is needed for internal concentrations to maximize at the target organ.

The use of continuous data test designs is not limited to toxicity studies but can be applied to other situations in which population assessments are made or biodiversity is measured. In these situations, it is important to select study sites that allow for exposures across a range of concentrations. Examples of such analyses include Adams et al. (2003), Ohlenforf (2003), and Beckon et al. (2008) on the effects of selenium on mallard duck egg teratogenesis. Well-designed studies of survival, reproduction, and other endpoints also can be used to predict the effects on wild populations (Hallam et al. 1989; Kooijman 1997) and likely would be useful for assessing the extent of injury and scaling restoration. We highly recommend test designs that result in continuous as opposed to discrete data sets to translate measurement endpoints to service loss.

Facilitating translation with data quality objectives

Application of a data quality objectives (DQO) process (e.g., USEPA 1994, 2000a, 2000b) can help ensure that information obtained in a site ERA also is compatible with the needs of the NRDA. A DQO process is a strategic planning approach based on the scientific method to prepare for a data collection activity. It provides a systematic procedure for defining the criteria that a data collection design should satisfy, including when to collect samples, where to collect samples, the tolerable level of decision error for the study, how many samples to collect, how the data will be interpreted, and attempts to balance risk and cost in an acceptable manner. In addition, the process will guard against committing resources to data collection efforts that do not support a defensible decision. The DQO process can be viewed as a means to bring disparate parties or stakeholders together to achieve a common objective with a common data set and decision criteria. The use of a DQO process leads to a robust set of data, decision criteria, and analysis output that allows for effective collection and assessment of data for both ERA and NRDA purposes.

Uncertainty in translation

Environmental management decisions typically are made within an analytical framework that includes varying degrees of uncertainty. Lack of data, extrapolation, variability within natural systems, and measurement precision are common sources of uncertainty that often impinges upon the decision-making process. The USEPA promotes the evaluation of uncertainty within ecological risk assessment and has issued a variety of policies and guidelines outlining methods for qualitative and quantitative consideration of uncertainty (e.g., USEPA 1997b, 1997c). In addition, a previous Pellston Workshop developed guidance for evaluating uncertainty in ERA (Warren-Hicks and Moore 1995). Within NRDA, Trustees commonly conduct uncertainty evaluations to support the development of injury estimates and restoration requirements.

Explicitly addressing uncertainty can substantially improve the range of options available for defining the scale of remediation and restoration. Analyses that present estimates of variance and ranges of plausible parameter estimates can characterize uncertainty and identify issues worthy of additional review or discussion. For example, it is often possible to identify factors that have a high degree of uncertainty but also have few implications for risk management decisions or damage calculations. Uncertainty analysis also can be used to define bounds on parameter estimates. These bounds might overlap with estimates developed by other parties, thereby creating an opportunity to build consensus on the basis of the uncertainty. Often, this approach is manifested through negotiation of exposure and effect scenarios that frame uncertainty within the study's data quality objectives (e.g., reasonable worst case scenario, 95% confidence interval, etc.).

In HEA, uncertainty often manifests in the translation of measurement endpoints into service losses. For example, laboratory studies for a single species of fish might correlate increasingly severe biological effects to increasing contaminant concentrations in sediment. The HEA practitioner might translate this series of data into a service loss function reflecting the degree of impairment observed across the range of contaminant concentrations observed at the site. Furthermore, these results might be applied to other fish species within the affected system. Unfortunately, defining specific service losses in this way has the potential to overstate the precision of the translation. Conducting bounding analyses on the laboratory-defined effect concentrations and the associated estimate of service loss will improve understanding of the range of uncertainty associated with the injury estimate, leading to better informed restoration decisions.

Whether qualitative or quantitative, expressing uncertainty when evaluating changes in ecosystem services can enhance environmental decision making. By expressing the degree of uncertainty associated with key factors, decision makers and analysts are more likely to focus on the issues of greatest relevance and eliminate factors that have little or no bearing on remediation or restoration decisions. In doing so, evaluations that express changes in ecosystem services offer a richer understanding of the range of likely effects and identify research opportunities that will ultimately reduce uncertainty in future assessments.

Biodiversity as a measure of ecosystem services

Measures of biological diversity are frequently included as endpoints in ERAs because of their perceived importance to ecosystems and society. Indeed, evidence is increasing that biodiversity directly influences the flows of ecosystem services. High species diversity maximizes resource acquisition across trophic levels and reduces the risk associated with stochastic changes in environmental conditions (Chapin et al. 1998). Conservation biologists have used the positive relationship between species richness (a measure of diversity) and ecosystem function to argue for greater species protection. The relationship between diversity and ecosystem processes is emerging as a fundamental concept in contemporary ecology. Although the specific shape of this relationship and its underlying mechanisms vary (Hooper et al. 2005), scientists and policymakers alike recognize the critical importance that species play in providing the goods and services that are essential for human welfare.

Here, we define biodiversity broadly to include aspects of genetic, species, and functional diversity within the spatial context of analysis. The positive relationship between species diversity and ecosystem function has been demonstrated in small-scale microcosms (Heemsbergen et al. 2004), marine tide pools (Bracken et al. 2008), large-scale field experiments (Tilman et al. 1997; Hector et al. 1999), and at a continental-global scale (Worm et al. 2006). The basic argument supporting this relationship is that greater species richness increases the likelihood that functionally important species will be present in an ecosystem. If we assume that these species have different functional roles and that the functions performed by any single species is limited, it follows that elimination of species will affect ecosystem processes. There is also evidence that greater diversity increases the resistance and resilience of ecosystems to anthropogenic perturbations (Frost et al. 1999). Most research on the diversity–ecosystem function relationship has focused on primary productivity, and the underlying mechanisms responsible for this relationship are generally well understood. Ecosystems with more species likely will use available resources more efficiently, resulting in greater primary productivity. In addition to productivity, a broad consensus within the scientific commu-

Table 1. Examples of ecosystem services that are directly related to species richness and diversity. Translation results were estimated on the basis of inspection of the relationships depicted in the original paper without considering uncertainty

Ecosystem service	Location and habitat type	Relationship	Translation	Reference
Primary productivity	Minnesota (USA) Grassland	Curvilinear	93% reduction in richness → 59% reduction in biomass	Tillman et al. (1997)
Productivity	8 European grasslands	Linear, log-linear	67% reduction in functional groups → 33% reduction in biomass	Hector et al. (1999)
Nitrogen uptake	California (US) grassland	Curvilinear	75% reduction in functional groups → 50% reduction in N uptake	Hooper and Vitousek (1997)
Decomposition and soil respiration	Soil microcosms	Linear	67% decrease in functional dissimilarity → 10% decrease in effect	Heemsberger et al. (2004)
Fish production	Open marine	Linear	71% reduction in richness → 80% reduction in average catch	Worm et al. (2006)
Nitrogen uptake	Marine tide pools	Linear	55% reduction in richness → 46% reduction in N uptake	Bracken et al. (2008)
Bioturbation	Marine benthic	Linear	99% reduction in density → 99% reduction in bioturbation	Solan et al. (2004)
Pollination	British fields	Unknown	60%–90% of species showed reduction in trait → 22% reduction in obligatory insect pollinated plants	Biesmeijer et al. (2006)

nity now is that species richness and functional diversity directly regulate many other ecological processes (Hooper et al. 2005), including nutrient dynamics, decomposition, soil respiration, and pollination (Table 1). Species loss within functionally related assemblages, such as pollinators and flowering plants, can affect ecosystem services at very large spatial scales (Biesmeijer et al. 2006).

Most studies investigating the relationship between diversity and ecosystem processes focus on a single trophic level; this relationship will certainly be more complex in systems with multiple trophic levels. Removal of species occupying higher trophic levels will have very different consequences for ecosystem processes compared with the loss of primary-level consumers. For example, species richness decreases at higher trophic levels and top predators are often more susceptible to anthropogenic disturbances. Consequently, an understanding of food web structure is necessary to predict the consequences of species loss on ecosystem function (Petchey et al. 2004). In systems regulated by top-down trophic interactions, removal of species at higher trophic levels would be expected to have greater effects (Downing and Leibold 2002).

Failure to consider the consequences of ordered compared with random species losses might cause analysts to underestimate the effects of species extinction on ecosystem function (Zavaleta and Hulvey 2004). Solan et al. (2004) compared the effects of species loss on ecosystem processes in marine sediments under random and nonrandom species extinction models. Removal of abundant, large, and highly mobile marine invertebrates had a much greater effect on ecosystem processes than did removal of smaller, less abundant species. Model simulations based on random and nonrandom extinction scenarios showed that the effects of species extinction on carbon storage were strongly influenced by the order in which species were removed (Bunker et al. 2005).

A good understanding of the underlying shape of the diversity–ecosystem function relationship is needed to translate loss of species diversity to reduced ecosystem services. Linear, curvilinear and threshold increases in ecosystem processes as a function of species diversity have very different implications (Figure 2). A linear relationship between ecosystem function and species diversity implies that all species are important and contribute equally to ecosystem processes (Figure 2A). A curvilinear relationship implies that some species are more important than others and that ecosystems could potentially loose a significant number of species without affecting function (Figure 2B). In a threshold functional relationship, ecosystem processes remain relatively saturated until species richness is reduced to a critical level,

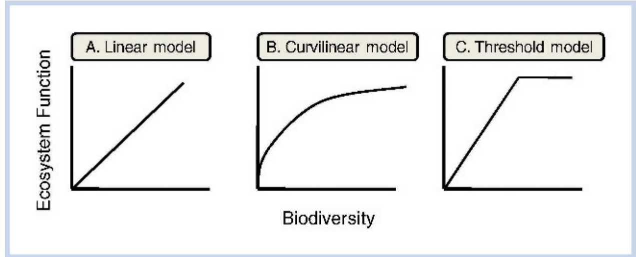

Figure 2. Linear (**A**), curvilinear (**B**), and threshold (**C**) relationships between biodiversity and ecosystem function.

causing a rapid decrease in ecosystem function (Figure 2C). Near this point, small changes in species diversity can produce large effects on ecosystem services. From a conservation biology perspective, a threshold relationship is probably of the most concern and forms the basis of the "rivet hypothesis" in ecology (Walker 1995). Similar to the rivets that attach wings to a plane, ecosystems are relatively resilient to species loss until some critical species (or rivet) is removed, causing catastrophic failure. Characterizing the nature of the relationship between diversity and ecosystem function and identifying the existence and location of any threshold would be useful for predicting ecosystem responses to species loss.

Relatively few studies have attempted to quantify the relative frequency of linear, curvilinear, or threshold relationships between diversity and ecosystem function. The examples shown in Table 1 include linear and nonlinear relationships. Hooper et al. (2005) concluded that the saturating response of ecosystem processes to increasing species richness was the most common pattern. However, Worm et al. (2006) found no evidence of functional redundancy in their global analysis of marine fisheries. An inconsistency obviously exists between the hypothesis that all species in an ecosystem are important and the alternative that ecosystems with a large number of species have significant functional redundancy. This inconsistency can possibly be resolved by considering functional traits of species instead of simple measures of species richness (Chapin et al. 1997; Heemsbergen et al. 2004). Indeed, some have argued that functional group diversity is a better predictor of ecosystem processes than is simple species diversity (Hooper and Vitousek et al. 1997; Tilman et al. 1997; Hooper et al. 2005). Regardless, these data strongly suggest a direct, quantitative relationship between species diversity and ecosystem processes.

We propose that measures of biodiversity can provide valuable insights about the risks to ESAEs and the service losses associated with contamination at hazardous sites. Although specific relationships between diversity and many ecosystem services remain to be described quantitatively, and recognizing that comprehensive studies to determine those relations are unlikely to be undertaken in most ERAs and NRDAs on a site-specific basis, the information provided in the studies cited above can form the foundation for reasonable translations. Additionally, high biodiversity has inherent aesthetic and cultural importance to some societies, facilitating quantification of the value of service losses and gains. We recommend that research attention be given to refining measures of biodiversity and their quantitative relationships with ESAEs for standard use in ERA and NRDA.

AGGREGATING SERVICE LOSSES

The NRDA attempts to consider the full range of injuries and service losses associated with site contamination. To ensure the comprehensive determination of damages, losses associated with multiple contaminants or involving multiple natural resources must be aggregated in ways that avoid under- or overestimation of those damages. The selection of aggregation approaches is often intertwined with the selection of a translation metric or metrics for relating measurement endpoints to service losses. Here, we review some of the approaches that can be used to aggregate service losses across multiple contaminants and across multiple resources. We begin by describing some recent case studies in which the aggregation problem is viewed as having been resolved successfully. These case studies also provide examples of how several measurement endpoints have been translated into service losses.

Recent case studies

Hylebos Waterway—The Hylebos Waterway in Commencement Bay, Washington, USA, is an industrial waterway contaminated with a wide range of organic and inorganic compounds. The waterway provides habitat for a range of species, including birds, resident and anadromous fish, shellfish, and benthic infauna (Commencement Bay Natural Resource Trustees 1991). To facilitate discussion regarding restoration of natural resources, the Trustees developed a HEA that assimilates impacts across contaminants and affected organisms. The Hylebos Waterway HEA offers 1 approach for translating contaminant impacts into service losses for selected species and then expressing these impacts on a habitat basis.

The underlying premise of the Hylebos Waterway HEA is that habitat is the appropriate currency for evaluating ecosystem function. To derive measures of habitat impacts, the Trustees evaluated contaminant-related injuries to the resource species that use the waterway by relating impacts to the degree of sediment contamination. This was accomplished by:

1. Dividing the waterway into a series of polygons and determining the habitat type and sediment contaminant concentrations in each polygon.
2. Assigning a habitat value to each polygon reflecting the baseline condition of the area and its ecological functions (e.g., as breeding habitat) related to juvenile Chinook salmon, juvenile English sole, and 4 bird assemblages.
3. Arraying published sediment effect thresholds for aquatic organisms for each contaminant or class of contaminants by concentration.
4. Assigning service losses on the basis of the expected severity of impact.
5. Determining for each polygon the service loss associated with each contaminant.
6. Aggregating service losses across contaminants to derive a single measure of service loss for all aquatic organisms in each polygon.
7. Calculating the service loss associated with each polygon in a given year by multiplying the aggregated service loss for aquatic organisms by the baseline habitat value.

Within this process are 3 key translations. First, the Trustees derived habitat values on the basis of the interaction of selected species with their habitats. However, it was necessary to translate each species–habitat combination into an aggregate measure of habitat value. The Trustees developed the aggregate value by weighting the individual scores for Chinook salmon by 50%, English sole by 25%, and the 4 bird assemblages by 25%. The increased weight for salmon in this application was based on the Chinook's status as a threatened species under the Endangered Species Act and on regional interest in restoration of the species (Commencement Bay Natural Resource Trustees 2002). The 2nd translation involved mapping the concentration of each contaminant to a service loss for aquatic organisms. This process was transparent and facilitated discussion among parties or highlighted opportunities to relate service losses quantitatively to empirically derived measurement endpoints. Finally, the Trustees translated the effects associated with individual contaminants into a single, combined measure of effect on aquatic organisms. This was accomplished through the concept of residual service loss. This concept expresses the effects of each contaminant in proportion to the sum of the service loss for all contaminants, wherein total service loss cannot exceed 100% of the habitat value. A complete description of the approaches used for all 3 translations is given by Commencement Bay Natural Resource Trustees (2002).

Lavaca Bay—Trustees worked cooperatively with Alcoa in Lavaca Bay, Texas, USA, to determine injuries to natural resources and resource services resulting from mercury and polycyclic aromatic hydrocarbon (PAH) releases from the Alcoa Point Comfort facility. A HEA framework was used to quantify service losses by habitat type. Using data from the RI/FS process, Trustees and Alcoa identified habitat types (such as emergent marsh, oyster reef, unvegetated subtidal soft bay bottom) and associated natural resources that had the highest potential to have been injured. Injuries to these habitats from contamination (and response actions) were quantified as degree of injury to key resource categories, including benthic invertebrates, finfish/motile shellfish (hereinafter referred to as "fish"), and birds, which were assumed to have suffered lethal (increased mortality) and sublethal (decreased fecundity, reduced growth, etc.) effects as a result of exposure to mercury or PAHs. A full description of the Lavaca Bay injury determination and restoration scaling is given at http://www.darrp.noaa.gov/southeast/lavacabay/admin.html.

This assessment was conducted with the use of a reasonable worst case (RWC) approach. The Trustees and Alcoa entered into a Memorandum of Understanding that specifically agreed to use data from the RI/FS to the maximum extent possible to determine natural resource injuries. Under the RWC approach, the Trustees agreed that before proceeding to plan and implement specific injury determination studies, the Trustees and Alcoa would consider existing data related to the affected area and the potentially affected resources. This included data from the RI/FS, historical data and results of scientific studies, and published literature reviews. With the use of this information, injury determinations were made by erring on the side of conservatism; that is, resource injury for an exposure level was assumed when at least 1 data source indicated adverse effects were reasonably likely. This approach is well suited for a cooperative assessment.

In the Lavaca Bay case, the cooperating parties evaluated site-specific sediment and tissue concentrations, compared these concentrations to threshold levels developed from onsite studies and the literature, and then determined service losses. The degrees of service losses were based on the type and severity of impacts associated with the measured tissue concentration ranges. Injury levels generally increased with increasing contaminant concentration, although no direct quantitative relationship was identified between effect and concentration. Specific assessments of service losses associated with 2 resource groups are highlighted below.

Benthic resource ecosystem services were identified as primarily relating to food production, decomposition, and energy cycling, which affect nearly all organisms within an estuarine system. It was assumed that impacts to benthic resources had the potential to impact biota in nearly all trophic levels. A goal of the assessment was to develop contamination concentration benchmarks for Hg and PAHs that are known or suspected injury thresholds for benthic resources on the basis of results from the RI/FS and general scientific literature. Data used from the RI/FS process included 1) site-specific chemistry to determine the nature and extent of PAH and Hg contamination in Lavaca Bay sediments (some additional NRDA-specific chemistry samples were collected to further refine the contamination characterization for PAHs); 2) laboratory bioassays and benthic macroinvertebrate studies (sediment quality triad) to determine relationships between mercury concentrations in field sediments and observed effects on survival, growth, and reproduction in benthic populations; and 3) habitat mapping that identified specific habitats important for benthos. Interim service losses were quantified on the basis of combinations of Hg and PAH concentration ranges and resulted in different levels of service reductions. The Trustees assumed the injuries resulting from Hg and PAHs were additive and assigned a level of injury, expressed as percent service loss, for each concentration range. The number of affected acres of benthic habitat at each injury level was determined through habitat mapping and contaminant sampling. This information was applied in the HEA to quantify total benthic injury.

The primary ecological services provided by fish in Lavaca Bay were identified to be food production for higher trophic levels and energy cycling. Injuries to fish were assumed to occur through direct exposure to Hg in sediments and surface waters and through ingestion of contaminated prey. Tissue data for fish and prey items collected as part of the RI/FS process, combined with literature studies that linked mercury tissue levels and adverse impacts, were evaluated with the use of a site-specific food web model. The food web model was designed to use selected species to represent major feeding guilds in Lavaca Bay. Similar to the benthic injury assessment approach, the Trustees and Alcoa used differing concentration ranges of Hg in fish tissues to derive the level of injury in fish. For each range of tissue concentrations, the Trustees determined a level of injury severity corresponding to service reduction percentage. The severity of adverse impacts to fish generally increased with increasing levels of fish tissue contamination. For ease of translating the service gains and losses in an HEA, injury to fish was determined on the basis of critical tissue concentrations as modeled from contaminated fringing marsh, vegetated and unvegetated, and oyster reef habitats (Evans and Engle 1994). Modeling was conducted as part of the RI process and applied to the injury determination. This allowed mapping of injured areas within Lavaca Bay on the basis of sediment Hg concentrations, which were associated with fish injury through modeling.

Southeast Texas sediment site—Sediment quality guidelines were used by Trustees to estimate losses of ecological services from the cumulative effects of PAHs and metals in intertidal sediments at a corrective action site in southeast Texas. Loss calculations were based on the probability of toxicity to marine amphipods according to the model developed by Long et al. (1998), which evaluates the incidence and magnitude of toxicity in sediments on the basis of the results of a 10-d toxicity test with marine amphipods. Mean effects range median quotients (Mean ERM-Qs) were developed for each sample by calculating the average of the ratios between the concentration of individual contaminants in sediment samples and their respective ERM values. The ERM values are numerical guidelines that are suggestive of adverse effects to sediment-dwelling organisms from exposure and bioaccumulation (Long et al. 1995). Adverse biological effects are highly probable at contaminant concentrations above the ERM. The

Trustees assumed that the sediment contaminants were available to sediment-ingesting organisms, and the Mean ERM-Q values for each location were then compared with the ranges of predicted toxicity established by Long et al. (1998). A direct translation was made between losses of ecological services provided by intertidal sediments and the predicted toxicity to marine amphipods. Sediments in locations with Mean ERM-Qs between 0.11 and 0.5, between 0.51 and 1.5, and greater than 1.5 were assigned service losses of 30%, 46%, and 74%, respectively, consistent with the predicted range of toxicities established by Long et al. (1998). The spatial extent of areas that contained Mean ERM-Qs within each of these ranges was then calculated.

Addressing multiple contaminants and natural resources

The primary goal for translating responses of ERA measurement endpoints to measures of service loss for an NRDA is to enable the Trustees to scale the amounts and types of restoration projects needed or damages assessed. In the simplest case, the effects of 1 contaminant on 1 resource species drive both the risk and damage assessments (e.g., effects of DDT on bald eagles). In such a case, a single translation from a measurement endpoint to a percent service loss can be relatively straightforward, and restoration projects can be scaled to that percent service loss. However, at most sites, multiple contaminants and multiple natural resources (e.g., species, guilds, communities, and habitat types) are present and interact. The degree of injury usually varies on the basis of individual and species-specific sensitivities to the contaminants and the varying concentrations of contaminants and times of exposure. To address this reality, service losses typically are translated into damages or restoration by aggregating across contaminants and across natural resources and developing a single or a few service loss translation metrics, or by aggregating across multiple service loss categories to produce the translation metrics. Each approach has advantages and disadvantages that should be considered on a site-specific basis.

Aggregating across contaminants—Several aggregation methods are possible when natural resources are exposed to multiple contaminants at a site. The measurement endpoint itself can be used to address a suite of contaminants. For example, a decrease in benthic invertebrate community diversity at a site relative to an appropriate reference site or a decrease in growth of test organisms in a sediment toxicity test could be used to quantify injury to the benthic community from a mixture of chemicals. Selection of these types of measurement endpoints for the ERA simplifies the translation to service losses.

Toxic equivalency approaches can be used to aggregate the effects of chemicals, especially for sites at which direct impacts to biota are not measured but are estimated by comparing concentrations of contaminants in abiotic media or tissues to effects levels from the literature. An assumption of additive toxicity has been used when the chemicals of concern share a common mechanism of action. This has been used most commonly at sites involving PCBs and other contaminants exhibiting Ah receptor–mediated toxicity and is the recommended approach for evaluating ecological risk from mixtures of PCBs, polychlorinated dibenzodioxins, and polychlorinated dibenzofurans (USEPA 2008). At sites with toxicologically dissimilar chemicals present, one possibility is to assume additivity, but this assumption should be evaluated as part of the uncertainty analysis. To illustrate, consider a situation in which the percent losses in reproduction expected from site concentrations of mercury, *p,p′*-dichlorodiphenyldichloroethylene, and copper are 30%, 50%, and 20%, respectively. Assuming these losses are independent and additive, the resulting toxic effect of these contaminants would result in only 7% residual reproductive services and a 93% decrease in reproduction with attendant service loss as:

$$(1-0.3)(1-0.5)(1-0.2)=0.07 \quad \text{and} \quad 1-0.07=93\% \tag{1}$$

The assumption that contaminants have additive effects could lead to either under- or overestimation of their true impact on reproduction, and the impact likely would differ depending on the species involved. Other scenarios and assumptions that could be postulated include:

1. Assuming that all of the toxic effects of the suite of contaminants is due to the single contaminant most toxic to that species, yielding a service loss as measured by reproduction of 20%, 30%, or 50%.
2. Assuming that all species are affected maximally by the most toxic contaminant, yielding a service loss as measured by reproduction of 50%.
3. Assuming an additivity ratio of less than 1, yielding a service loss as measured by reproduction between 20% and 93%.
4. Assuming some degree of synergism or antagonism among contaminants, yielding a service loss as measured by reproduction of potentially less than 20% or greater than 93%.
5. Assuming that losses are species specific, the magnitude of impact varies as a function of other environmental conditions during the time of exposure, and the concentration of the contaminants is not constant through time.

To address all scenarios would be time consuming and costly, and it is as likely as not that the results of at least some of the tests would be equivocal. Therefore, we strongly support thorough uncertainty analyses to identify the most important variables affecting estimates of service losses, together with statements that serve to bound the range of possible losses.

In some situations, either the suite of contaminants and potentially impacted natural resources are few, or the amount of empirical information available is relatively large. For example, at sites where sediment toxicity is the primary concern, large empirical databases of effects ranges are measured for exposure to multiple chemicals at environmentally relevant exposures. These databases can be mined to establish an appropriate measurement endpoint (e.g., probability of toxicity or exceeding a Sediment Quality Guideline) for translation to service loss. This type of approach, described above, was used by NRDA practitioners at the Southeast Texas sediment site to estimate service losses from the cumulative effects of PAHs and metals in intertidal sediments. In that case, Mean ERM-Qs were calculated across chemicals and related to a service loss on the basis of the underlying probability of toxicity. Rather than using a mean quotient, Field et al. (2002) proposed using the maximum probability of observing a toxic response (P_{Max}) in sediments as an estimate of service loss. P_{Max} is derived from the maximum probability of toxicity across logistic regressions of probabilities of toxic responses for individual chemicals. The individual chemical logistic regressions used to derive P_{Max} come from a large database of environmentally relevant sediment toxicity information. Field et al. (2002) posit that P_{Max} has certain advantages over the averaging approach, but because both cases rely on literature-derived data, sites with unique resource species, chemical mixtures, or mixture ratios could require site-specific information.

Aggregating across natural resources—At most NRDA sites, multiple natural resources suffer injury, and the total service loss needs to be translated as 1 or more metrics to scale the appropriate amount of restoration. This is often done by aggregating the total service loss into a loss of habitat and by scaling that loss to an equal amount of habitat restoration through the HEA process. Aggregation of service losses can be addressed at more than 1 point in a HEA, and several different approaches have been used to develop translation metrics to aggregate measurement endpoint responses or service losses across multiple natural resources to guide selection of restoration projects.

One approach is to develop a single function of total residual services relative to exposure concentrations (Cacela et al. 2005). In this approach, all relevant toxicity endpoints and dose–response relationships over a variety of affected resources are considered by an expert panel. The panel relates the available toxicity information and exposure levels to the residual level of services and then multiplies the residual services at enough points across the range of exposure concentrations to develop a relationship between exposure concentration and total residual services. An underlying assumption in this approach is that the available toxicity relationships include an appropriate distribution of natural resources so that natural resource services are represented fully. This approach likely is more appropriate in cases in which the adverse effects on all relevant natural resources occur over a similar range of exposure conditions than it is in cases in which responses vary significantly among natural resources. The total residual services might be underestimated if too many similar residual services are aggregated in the multiplicative function. In the illustration provided by Cacela et al. (2005), the goal was to estimate service losses relating to PAHs in the sediments that were thought primarily to affect fish and benthic invertebrates. In this case, the adverse effects threshold data and the exposure–response curves for both impacted groups covered the same general range of sediment PAH concentrations (1–250 mg/kg dry weight).

Another approach for aggregating service losses across multiple natural resources in a habitat is to estimate losses to different categories of natural resources separately and then to aggregate losses across categories. In this approach, practitioners select several categories of natural resources, assign a fraction of the total resource services to each category, develop translation metrics within each category, calculate a percent service loss within each category, and then calculate a total service loss with the use of the weighted sum of percent service loss across the categories:

$$\text{Service loss} = \sum (\text{fraction of service for resource category } i) \times (\text{service loss for category } i) \qquad (2)$$

An approach like this was used for the Hylebos Waterway described earlier. One advantage of this approach is that it allows different types of toxicity data and translation metrics to be used for different categories of organisms, thus avoiding the need to combine dissimilar types of information. This approach can result in apparently illogical conclusions, however, if 1 category of natural resources is significantly more affected than the others. To illustrate with a hypothetical example, assume that marsh birds are determined to represent one-fifth of the service flows in a given marsh habitat and are severely affected, whereas the natural resources making up the other 80% of the service flows are not directly affected. In this case, the maximum percent service loss for the habitat would be only 20%, even if the birds were completely extirpated at the site. In this situation, the Trustees could conduct an REA for the birds and determine the amount of habitat needed to create the appropriate number of discounted bird years in lieu of the HEA.

Another alternative would be to evaluate natural resources in categories and then aggregate across categories in the restoration portion of the HEA, rather than to attempt to estimate a total service loss. In this approach, practitioners would select several categories of natural resources, develop translation metrics within each category, calculate a percent service loss and corresponding DSAYs for each category, and then look for restoration projects that address all of those DSAYs at once. For example, consider a situation involving excessive selenium in a tidal marsh. Toxicity thresholds for effects on vegetation and benthic invertebrates and a dose–response curve for marsh birds could be used separately to translate the toxicity information for each natural resource category into service losses. The result might suggest that the vegetation loss would require 5 acres of tidal marsh, the benthic invertebrate loss 15 acres of tidal marsh habitat, and the bird loss 85 acres of tidal marsh. In this situation, a single restoration project that creates and maintains 85 acres of tidal marsh would address all of the injury categories simultaneously.

Ultimately, the advantages and limitations of any aggregation approach need to be evaluated relative to the facts and circumstances of each NRDA case. The use of expert panels or cooperative assessments can be helpful in determining the number of contaminants and natural resources or resource categories to be evaluated, the translation relationships to be used, and the aggregation approach that is appropriate to ensure that the most logical and transparent restoration scaling approach is identified. The potential impact of the uncertainty introduced in each step should be carried through the analysis to estimate its impact on the range of sizes and costs of restoration projects. In this way, the parties should be able to determine whether the existing uncertainties can be addressed through restoration (especially since larger restoration projects are generally more cost effective than smaller ones on a per acre basis) or whether additional study might be warranted to reduce specific uncertainties in the analyses.

SUMMARY OBSERVATIONS

The process of translating measurement endpoints from ERA and other assessments conducted as part of a CERCLA RI/FS into service losses for NRDA injury determination and restoration scaling is complex. Although there is not full agreement among Trustees, industry representatives, USEPA, consultants, and academics, the authors of this paper believe that both ERA and NRDA would benefit generally if ecosystem services provided by natural resources to humans and ecosystems are used as a common currency. We also believe that no single method or approach for translating measurement endpoints into service losses, or for aggregation of those losses, is applicable to all situations and sites. Selection of the most appropriate translation or aggregation approach undoubtedly will involve negotiations among the various parties involved at a site and will incorporate many of the considerations identified in this paper.

The types of data required by ERA and NRDA generally are very similar, although the measurement endpoints themselves often vary. Consequently, integration of ERA and NRDA could be enhanced through the use of resource- or service-related assessment and measurement endpoints (the common currency) in the ERA that would both benefit the ERA and better inform the NRDA. We recommend that Trustees, BTAG members, and other parties with expertise in ERA and NRDA convene to expand the set of the USEPA's (2003) GEAEs to include additional ecosystem service endpoints that are particularly responsive to the needs of both assessments at local and national scales.

Enhanced integration of assessment approaches likely will require involvement of NRD practitioners representing the Trustees and responsible party or potentially responsible parties (RP/PRPs) early in the planning and design of the ERA, nature and extent determination, and other pertinent components of the remedial investigation. This would enable incorporation of service-based assessment and measurement endpoints into the ERA that could readily be translated into service losses for NRDA purposes. Supplemental data collection also could be incorporated into the nature and extent component to better inform both the ERA and damage assessment. Recognizing that incorporation of additional NRD-related endpoints or data collection into an RI is beyond the USEPA's regulatory authority, it is our view that most RP/PRPs would be willing to provide the additional funding, generally via a specific funding and participation agreement, in that it would increase efficiency, reduce overall costs, and promote a quicker and more satisfying decision process.

Integration might best be facilitated with a DQO process to determine modifications or enhancements to traditional ERA measurement endpoints or to identify additional studies that would be required for injury determination and restoration scaling. The DQO process also would ensure that endpoint changes do not compromise the ERA. Involvement of the BTAGs could assist in the integration process. Traditionally, BTAGs have expressly avoided discussing NRDA-related matters; we believe this has lead to inefficient and more costly remedial and restoration actions than could be achieved by a more integrated approach.

Biodiversity measurement endpoints were acknowledged to be good indicators of many important ecosystem services. We recognize, however, that quantification of the relationships between biodiversity and ecosystem service risk and losses could require substantial additional investigation, something not likely to be supported in most CERCLA cases. Consequently, we encourage continued attention by the environmental and ecological research communities to help establish the linkages between diversity measures and ecosystem processes for use in ERA and NRDA.

Finally, it is our opinion that methods for aggregating service losses should be selected on a site-specific basis and that no best approach exists universally. Examples of 3 approaches that we believe to be particularly useful are provided, together with the discussion of potential methods for aggregating across contaminants and across resources to aid practitioners in determining the method that is most applicable for their sites. We encourage further innovation in the development and evaluation of aggregation methods to enhance quantification of ecosystem service losses.

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Disclaimer—Although some of the authors of this paper are employees of governmental agencies, the ideas described herein do not necessarily reflect the policies of those agencies, and no official endorsement should be inferred. Mention of trade names or commercial products does not constitute endorsement or recommendation for use.

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