

Ecological Risk Assessment and Natural Resource Damage Assessment: Synthesis of Assessment Procedures

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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

The Society of Environmental Toxicology and Chemistry (SETAC) convened an invited workshop (August 2008) to address coordination between ecological risk assessment (ERA) and natural resource damage assessment (NRDA). Although ERA and NRDA activities are performed under a number of statutory and regulatory authorities, the primary focus of the workshop was on ERA and NRDA as currently practiced in the United States under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). This paper presents the findings and conclusions of the Synthesis Work Group, 1 of 3 work groups convened at the workshop. The Synthesis Work Group concluded that the different programmatic objectives and legal requirements of the 2 processes preclude development of a single, integrated ERA/NRDA process. However, although institutional and programmatic impediments exist to integration of the 2 processes, parties are capitalizing on opportunities to coordinate technical and scientific elements of the assessments at a number of locations. Although it is important to recognize and preserve the distinctions between ERA and NRDA, opportunities for data sharing exist, particularly for the characterization of environmental exposures and derivation of ecotoxicological information. Thus, effective coordination is not precluded by the underlying science. Rather, willing participants, accommodating schedules, and recognition of potential efficiencies associated with shared data collection can lead to enhanced coordination and consistency between ERA and NRDA.

Keywords: Ecological risk assessment Natural resource damage assessment CERCLA Assessment endpoints Hazard quotient

INTRODUCTION

The Society of Environmental Toxicology and Chemistry (SETAC) convened a workshop to address perceived and real difficulties in coordinating or harmonizing the practices of ecological risk assessment (ERA) and natural resource damage assessment (NRDA). Although ERA and NRDA activities are performed under a number of legal and regulatory authorities in the United States, the primary focus of the workshop was on ERA and NRDA as currently practiced under the Comprehensive Environmental Response Compensation and Liability

Act (CERCLA 1980). The ERA process systematically evaluates how likely it is that adverse ecological effects might occur as a result of exposure to 1 or more stressors (USEPA 1998). Ecological risk assessment can be prospective (prediction of the likelihood of future effects) or current (evaluation of the likelihood that observed effects are associated with current exposure to stressors). Natural resource damage assessment is a process by which injuries (i.e., measurable adverse changes) to natural resources are determined and quantified for purposes of establishing damages. Natural resource damage assessment is current, retrospective, and prospective, in that damages can be sought for natural resource injuries that are occurring, have occurred in the past, and are reasonably expected to continue in the future.

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In the workshop, similarities and differences in the ERA and NRDA processes were discussed. In this manuscript, we report the findings and conclusions from the Synthesis Work Group. We focus on the common and disparate elements of ERA and NRDA from assessment design through risk characterization or injury determination. The manuscript is organized to first highlight the objectives of the ERA and NRDA processes. Understanding these distinct statutory/regulatory objectives is fundamental to understanding the sometimes divergent scientific approaches that are relied upon in the 2 processes. Next, common steps in the design and framing of risk and injury assessment are compared and contrasted, including selection of assessment endpoints and response measures, estimation of exposure, spatial and temporal scope of assessment, use of background (reference) and baseline conditions, derivation and application of adverse response thresholds, and use of hazard quotients to characterize risk or injury. For each of the assessment steps, impediments to coordination are discussed and recommendations to improve coordination are provided.

Ecological risk assessment and NRDA share many common types of data needs and analytical constructs; however, the unique programmatic objectives and statutory/regulatory requirements of each can give rise to divergent data requirements and analytical approaches. Consequently, development of a single, integrated methodology that encompasses both ERA and NRDA programmatic needs is neither practical nor desirable. Nonetheless, opportunities exist for coordination of aspects of the 2 frameworks.

ERA AND NRDA: DIFFERING PROGRAMMATIC OBJECTIVES

Both ERA and NRDA (specifically, the injury assessment component of NRDA) consider and evaluate adverse effects of hazardous chemical exposure on ecological resources and ecosystem processes. However, the 2 assessment programs have different programmatic and scientific objectives that derive from their respective statutory and regulatory authorities and guidance. In the context of CERCLA, ERA is performed to inform response/remedial decision making. Natural resource damage assessment, on the other hand, is aimed at compensating the public for injury, destruction, or loss of natural resources. Compensation in NRDA is achieved through the restoration, rehabilitation, replacement, or acquisition of natural resources.

As a consequence of these distinct objectives, ERA seeks to answer fundamentally different questions from NRDA. Ecological risk assessment is focused on aiding remedial decision-makers in evaluating whether, and what, actions should be undertaken to manage risks to the environment. An ERA is therefore sufficient when it provides adequate information to support such decisions. Natural resource damage assessment, in contrast, is focused on quantifying the compensation necessary to restore injured resources to baseline and to offset past and future injuries to natural resources. An NRDA is therefore sufficient when adequate information is provided to support determinations regarding the nature and extent of natural resource injuries and to quantify the compensation required to offset losses to natural resources and their services.

Risk and injury

These programmatic distinctions between ERA and NRDA are reflected in the definitions and interpretations of the

terms “risk” and “injury.” The *Webster's New Collegiate Dictionary* (Webster 1979) defines risk as the probability of an adverse consequence. The U.S. Environmental Protection Agency (USEPA) defines risk as the expected frequency or probability of undesirable effects resulting from exposure to known or expected stressors and the likelihood that adverse ecological effects might occur or are occurring as a result of exposure to 1 or more stressors (USEPA 1998). In general use, “injury” is defined as damage or harm done to or suffered by a person or thing (American Heritage Dictionaries 2000). As employed in NRDA, “injury” has been defined in the US Department of the Interior (USDOI) regulations as a measurable adverse change in natural resources resulting either directly or indirectly from exposure to a discharge of oil or release of a hazardous substance. Specifically, injury means “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” (USDOI 2005a, 43 CFR §11.14[v]). As defined by NOAA's NRDA regulations for assessment performed pursuant to the Oil Pollution Act, “Injury means an observable or measurable adverse change in a natural resource or impairment of a natural resource service” (US Department of State 2005, 15 CFR §990.30).

As highlighted by general definitions, risk incorporates the concept of frequency and likelihood of occurrence. As commonly practiced at contaminated sites, ERA often focuses on evaluating the reasonable likelihood (or potential) that an adverse effect has or would occur, typically over some area of concern. Less commonly, ERAs quantify the severity, duration, permanence, or probability of adverse effects. In contrast, the definition of injury adopted in NRDA does not explicitly include the concept of potential loss or harm. Rather, natural resource trustees must demonstrate the occurrence of injuries to natural resources, not just the potential for adverse effects, and a connection between the injury and a constituent in the release. These injuries then serve as the basis for quantifying damages. The NRDA process necessarily considers the severity, spatial extent, and temporal extent of injury to calculate appropriate measures of compensation.

Protectiveness and burden of proof

Another consequence of the distinct objectives of the 2 programs relates to the concept of “burden of proof.” Because risk assessment is designed to guide risk management and remedy selection, it often includes assumptions intended to ensure that response actions implemented are protective of environmental receptors. However, determination of a specific severity or frequency of adverse effect might not be required. Furthermore, risk management involves tradeoffs between the precision and expense of analysis and its relevance to decision making. In the context of CERCLA response actions, a relatively high degree of uncertainty in risk estimates might be acceptable, depending upon the nature of the decision to be made.

In contrast, NRDA is compensatory. Natural resource damage assessment therefore requires determinations of the nature and extent of adverse changes in the chemical or physical quality or viability of a natural resource, not just the probability or likelihood of such adverse changes. The burden of proof in NRDA is based on the Trustees' obligation to

determine and quantify injured natural resources for purposes of quantifying the appropriate level of damages.

Overall, ERA is designed to inform response/remedial decisions; NRDA is designed to enable Trustees to seek compensation for the public to offset past and future natural resource losses. Whereas ERAs can conclude with the determination of potential ecological risks, NRDA is designed to determine and quantify measurable injuries and service losses, a bar that is often higher than determining whether a risk of injury exists. Consequently, the information developed in an ERA generally is not sufficient for a complete injury assessment or quantification of damages as part of an NRDA.

ERA AND NRDA: TECHNICAL SIMILARITIES AND DIFFERENCES

Despite the programmatic distinctions discussed above, ERA and NRDA share several scientific elements. Understanding areas of commonality, as well as process-related distinctions in how common elements are implemented, will assist practitioners in identifying opportunities for efficient data sharing.

Common elements

Figure 1 illustrates some of the conceptual commonalities between ERA and NRDA. Both ERA and NRDA entail the identification and selection of assessment endpoints and response measures. Selection includes consideration of the level of biological organization (e.g., suborganism, organism, population, community), the specific receptor to be assessed, and the adverse responses to be evaluated. Both processes then proceed to an evaluation of risk or injury. That evaluation typically is based on characterizing site conditions and understanding the nature of receptor exposure to hazardous substances, establishing relevant adverse response thresholds, and determination and often quantification, of risk or injury. Despite these conceptual similarities, however, ERA and NRDA often employ divergent approaches that are, in part, a function of the differing programmatic objectives and requirements noted previously. Below, we discuss several of the conceptually shared elements of ERA and NRDA.

Assessment endpoints and response measures

Both ERAs and NRDA have been undertaken with the use of assessment endpoints and biological response measures at multiple levels of biological organization (Figure 2). Suter et al. (2005) provides definitions of the various levels of biological organization; this concept is also discussed by USEPA (1998, 2003). Therefore, opportunities for coordination in the selection of assessment endpoints and response measures should be explored in the planning and conduct of ERA and NRDA, although it should be recognized that different endpoints and measures may be selected.

For ERAs at contaminated sites, the population is the most commonly targeted level of biological organization for development of assessment endpoints. The USEPA (1998) defines a population as “an aggregate of individuals of a species within a specified location in space and time.” Population-level attributes (e.g., abundance, production, extirpation) can be measured directly in fish, amphibian, avian, and mammalian receptors evaluated in ERAs conducted at large sites. Community-level endpoints (e.g., benthic invertebrate surveys, fish surveys) are employed relatively routinely in ERA, particularly in aquatic systems. In some instances, USEPA recognizes the

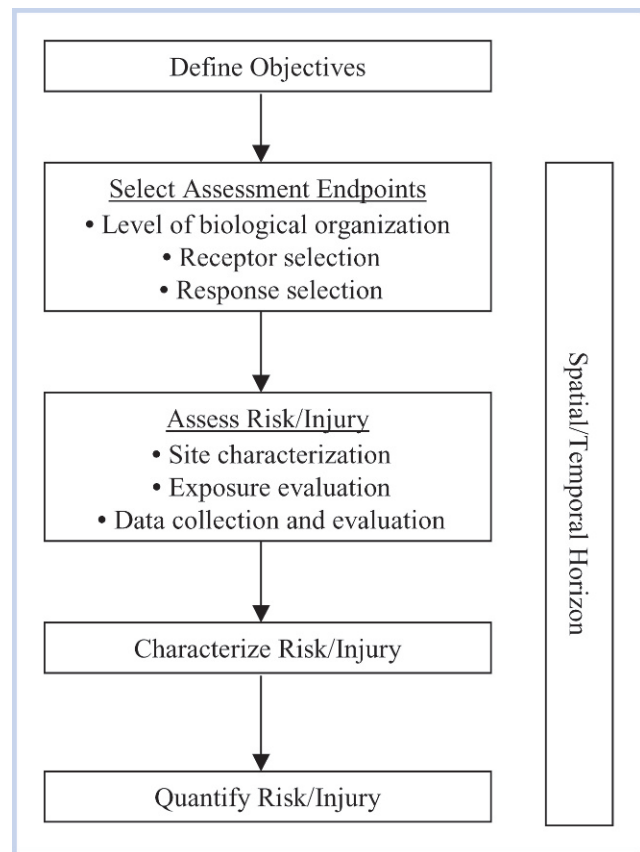


Figure 1. Conceptual similarities between ecological risk assessment (ERA) and the injury assessment phase of natural resource damage assessment (NRDA).

importance of protecting individual organisms—particularly special-status species, and organism-level endpoints and response measures are employed in these settings. For example, protection of the individual is mandated by the Endangered Species Act, Marine Mammal Protection Act, Bald Eagle Protection Act, and Migratory Bird Treaty Act (USEPA 1998). Suter et al. (2005) notes that, in practice, most ERAs focus on organism-level attributes of a population, and organism-level attributes (survival, growth, reproduction) commonly are used to infer population-level risks.

Compared with ERA, biological response measures are evaluated over a wider range of levels of biological organization in NRDA, although this represents a point of contention between trustees and responsible parties (who generally have favored a population/community approach). Natural resource damage assessment injury determinations commonly involve biological responses at the suborganism (e.g., enzyme induction, physiological change) and organism level (MacDonald et al. 2002; Cacula et al. 2005). Indeed, the USDOJ NRDA regulations specifically identify a number of suborganism and organism biological responses as meeting regulatory definitions of injury. For example, USDOJ (2005b, 43 CFR §11.62[f][1][i]) defines injury occurring if a biological resource or its offspring have “undergone at least one of the following adverse changes in viability: death, disease, behavioral abnormalities, cancer, genetic mutations, physiological malfunctions (including malfunctions in reproduction), or physical deformations.” The regulations define fish neoplasm as an injury “when a statistically significant difference can be measured in the frequency of occurrence of the fish neoplasia when comparing

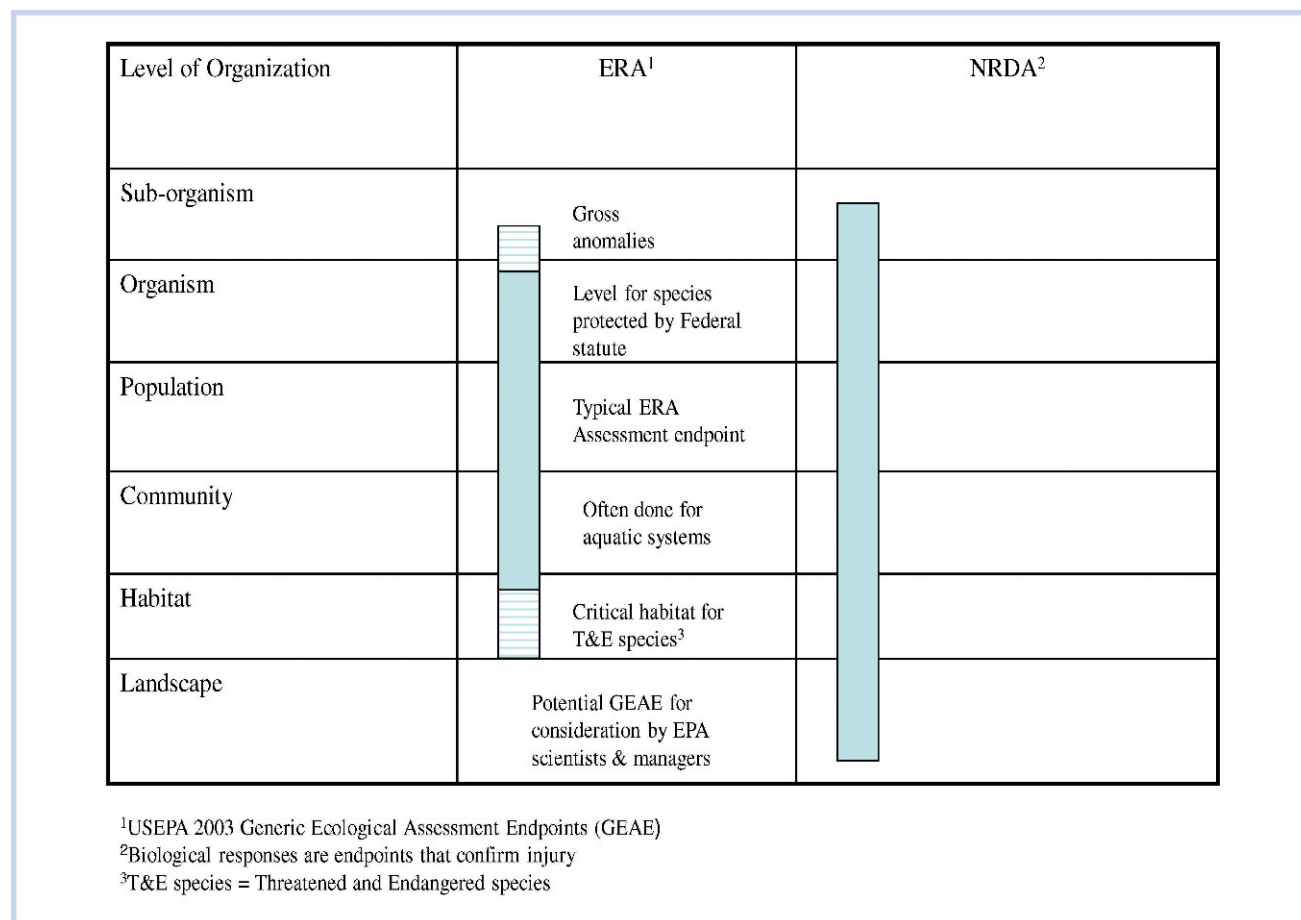


Figure 2. Levels of biological organization commonly employed in ecological risk assessment (ERA) and natural resource damage assessment (NRDA). Dark shading indicates levels of organization that are employed relatively frequently. Lighter shading indicates levels of organization less commonly employed.

population samples from the assessment area and a control area.” Other examples of suborganism endpoints in the USDOJ NRDA regulations include measurements of eggshell thinning, cholinesterase enzyme inhibition, delta-aminolevulinic acid dehydratase (ALAD) inhibition, and physical deformities (including external malformations, skeletal deformities, whole organ and soft tissue malformation, and histopathological lesions). Suborganism measures are not common in ERA, at least in part, because these sublethal measures have weaker causal links to organism processes of growth, survival and reproduction (Tannenbaum 2005; Emlen and Springman 2007).

In addition to the organism-, population-, and community-level endpoints and response measures common with ERA, NRDA also assess habitat-level response measures, such as a reduction in the area or quality of habitat from the time of injury until the resource recovers to baseline, especially when habitat equivalency analysis (HEA) is used to scale restoration (NOAA 1997; Dunford et al. 2004). Habitat-level endpoints and response measures are less commonly used in ERA. USEPA (2003) suggests the use of habitat-level assessment endpoints (area and quality) only when contamination is present in critical habitat for Special Status species, although some states (e.g., Massachusetts) require evaluation of wetland habitat as part of the ERA.

As the discussion above illustrates, an area of commonality does exist between ERA and NRDA when considering certain levels of biological organization. The focus on organism, population, and community endpoints and response measures

in both ERAs and NRDA could therefore present the greatest opportunity for common data to be collected and analyzed. On the other hand, because suborganism and habitat measures in NRDA are less commonly employed in ERAs, there is less chance that ERAs will collect usable data for NRDA at these levels of biological organization. A companion paper (Munns et al. 2009) encourages the consideration of an ecological services (habitat-level) endpoint for ERAs that might enable greater coordination of ERA and NRDA in the area of assessment endpoints and response measures.

Receptor selection

Both ERAs and NRDA require the selection of receptors, variously referred to as indicator species, receptors of interest, receptors or resources of concern, representative species, or representative natural resources. The simple term “receptor” is used here to refer to the species or environmental media (e.g., groundwater, surface water) that are evaluated in an ERA or NRDA. The term “receptor” does not occur in the USDOJ NRDA regulations, which refer to natural resources (e.g., surface water, fish, wildlife). In ERAs, receptors are generally selected to represent the major feeding guilds or trophic levels at a given site (e.g., the benthic invertebrate community; fish populations; wildlife species representing piscivores, omnivores, and/or invertivores). Receptors often are chosen to be among the most susceptible (i.e., most highly exposed and most toxicologically sensitive; USEPA 1998) of the species likely to inhabit a given site (i.e., the risk drivers), with the assumption that extrapolation of risk conclusions regarding these receptors

are protective of other, less susceptible species. Surrogate species can also be selected in ERA on the basis of data availability (e.g., although green herons might forage at the site, great blue herons may be selected as a surrogate species because of greater availability of data on life history and ecotoxicity). Natural resource damage assessments, however, are less likely to select purely surrogate species and are more likely than ERAs to select multiple representatives of each resource guild because of the trustees' need to determine and quantify injuries to natural resources in the assessment area.

It is no surprise then that ERAs and NRDA's conducted at the same site might select receptors that only partially overlap, given their inherently different objectives—namely risk management decisions, as opposed to compensation for injury to natural resources. Ecological risk assessments must ensure that the most susceptible species within each relevant feeding guild—that is, the likely risk drivers—are represented so the remedy that is selected is protective of all species likely to inhabit the site. In contrast, NRDA receptor choices revolve around the trustees' need to determine and quantify the spectrum of injuries to resources present at the site to ensure adequate and effective compensation (e.g., restoration). One opportunity for coordination and cooperation between ERA and NRDA in receptor selection could lie in the early involvement of trustees during ERA problem formulation, to explore whether the receptors selected for ERA will also support NRDA data needs.

Exposure analysis

The accurate estimation of chemical exposure from the release of hazardous substances to the receptor is a fundamental aspect of both risk and injury analysis. Often the measurement used to reflect such exposure in ERA is the estimated environmental concentration (EEC).

Multiple tools for estimating chemical exposures can be used, ranging from simplistic deterministic characterizations of EECs (e.g., maximum value, upper confidence limit of the mean exposure [95 UCL]) for the various exposure media (soil, surface water, sediment, biota) to more complex probabilistic characterizations (e.g., Monte Carlo analysis, geostatistical/geoanalytical tools [kriging, Thiessen polygons]). Estimated environmental concentrations for receptor-specific exposure areas or individual point estimates are then compared with toxicity thresholds (most common for ERA) or dose–response relationships (often used in NRDA, less often employed in ERA) to evaluate the potential for risk and injury determination. Ecological risk assessments focus on estimating current and anticipated postremediation exposures. Natural resource damage assessments generally include determination of past and future exposure in that past and future injuries are included in injury and damage quantification.

Substantial opportunity for coordination between ERA and NRDA exists (and often occurs) for characterizing chemical exposure to receptors, although, as discussed below, the spatial and temporal scope of such characterizations might not wholly overlap. The obvious benefit of such coordination is a reduction of redundant data collection and chemical analyses and more cost-effective and timely exposure analysis.

Spatial and temporal scope

A fundamental distinction in the design and conduct of assessments in ERA and NRDA is the spatial and temporal domain of analysis. CERCLA ERAs are components of

remedial investigations (RIs), which are designed to inform remedial decision making by identifying areas of contamination, if any, that exceed a designated threshold or otherwise pose an unacceptable risk. Generally, the designation of areas of unacceptable risk is more important than the magnitude of the risk, how long the risk existed, or how long the risk will remain after the completion of remedial action. After risk analysis, remedial alternatives for mitigating the identified unacceptable risk are evaluated. As a practical matter, however, quantification and delineation of all areas that pose unacceptable risk might not be needed to reach a final decision regarding remediation. In addition, ERAs often use multiple response measures and receptors (i.e., multiple lines of evidence) for an area of contamination to improve the confidence of the determination of acceptable or unacceptable risk, rather than determine the need for independent (and likely duplicative) remedial action to address the unacceptable risk associated with each individual response measure and receptor.

Natural resource trustees, on the other hand, seek to understand the spatial extent of contamination and natural resource injuries if the public is to be fully compensated for losses to trust resources. Measurable adverse effects and resultant injury might occur at levels deemed “acceptable” (i.e., not subject to remedial actions) to risk managers. In such instances, the spatial scope of the NRDA could extend beyond the study area considered in an ERA. In addition, trustees also might need to investigate gradients of exposure and injury from the source of contamination to determine the extent and magnitude of injury or service loss. Each resource category (and “receptors” within each resource category) could have a different spatial extent of injury that would need to be characterized independently. Thus, assessment of the spatial extent of resource injuries will often entail more data and analysis than the spatial extent of ecological risk performed in an ERA.

The temporal domain of ERAs and NRDA's also differ in that NRDA practitioners assess injury and damages in the past, present, and future—spanning from a specified historical point in the past (or the time of the release, or both) into the future, until the injured natural resource and the services it provides return to baseline conditions. CERCLA ERAs typically are focused on evaluating current risks and less often future risk (i.e., post-remediation, if reasonable future land use might lead to increase exposure and risk). Natural resource damage assessments typically require a broader suite of data inputs and further analysis than is used in most ERAs because of the need in NRDA to evaluate conditions in the past, as well as projecting future injuries (especially quantification of impact–recovery curves).

For determination of current and future risk/injury, there is opportunity to coordinate and share spatial and temporal information between ERA and NRDA. However, to determine the full extent of natural resource injuries and service losses, the spatial description of injuries in the NRDA often includes a description of the continuum of injury assessed in a comprehensive spatial and temporal context. Consequently, the spatial and temporal characterization of injury in NRDA is often more complex than is employed in ERA.

Background versus baseline

In evaluating adverse effects of chemical contaminants on ecological receptors, ERAs often include a comparison of site data with background (reference) information intended to

provide a reasonable expectation of conditions that would be expected at the site but for releases of hazardous substances (USEPA 1994, 2002). Background information may be obtained from nearby locations that are ecologically similar to the contaminated site, from prerelease historical data (if available), or from statistical approaches (USEPA 1994). In selecting background locations, USEPA (1994) recommends consideration of factors such as the similarity of physical, chemical, and biological characteristics. Although not necessary to determine ecological risk from chemical exposure, chemical concentrations at background locations are used in ERA to identify site-related chemicals (USEPA 2002), and background response levels (e.g., reference growth and survival in sediment toxicity tests) are used to develop the threshold for unacceptable risk (i.e., significant difference from background response; Long and Chapman 1985; Menzie et al. 1996).

“Baseline” in CERCLA NRDA is the condition that would have existed in an assessment area had a release of hazardous substance not occurred (USDOJ 2005a, 43 CFR §11.14[e]). As with ERA, NRDA baseline entails consideration of physical, chemical, and biological factors that might influence baseline conditions (USDOJ 2005c, 43 CFR §11.72). Baseline conditions in NRDA are determined to establish the amount of injury and service loss (in different locations and at different points in time) that resulted from the release. The loss is equal to the difference between the injured state and the baseline state. Thus, baseline typically will be defined for all resource categories and response measures (Cacela et al. 2005). Although analogous to the use of background concentrations and response levels in ERA to assess unacceptable risk, ERA often makes a binary determination of acceptable (reference) or unacceptable response; rarely is the magnitude of the difference from the reference response used to quantify the severity of risk.

Another difference between background and baseline is that in NRDA, baseline conditions include consideration of anthropogenic factors that can influence environmental conditions (e.g., the presence of dams, land uses, and habitat quality; Barnthouse and Stahl 2002; Burger et al. 2007). Thus, baseline encompasses all factors, natural and anthropogenic, that might influence the services provided by a biological resource. As a result, information needs for NRDA baseline determination often are greater than the requirements for background determination in ERA.

Although opportunities for coordination related to background and baseline information exist (e.g., selection of reference locations), characterization of background conditions in ERA often will be insufficient to meet the needs of baseline determination for NRDA.

Derivation and application of adverse response thresholds

Ecological risk assessment and NRDA both rely on the derivation and application of adverse response thresholds to inform determinations of risk or injury. The basic tools used to develop these response thresholds are likewise shared by the 2 processes and derive from the field of ecotoxicology. Potential ecological effects associated with a chemical stressor in the environment can be evaluated with the use of a spectrum of tools, ranging from literature-based toxicity information to site-specific studies. However, as with the other assessment components discussed above, specifics of threshold derivation and application might differ between ERA and NRDA because the uses of the information (whether literature-based or site-specific) reflect differing

programmatic objectives. For example, whereas ERAs generally use adverse effect thresholds to delineate areas of concern sufficient to inform risk-based decision making, NRDA injury assessments often will use the underlying ecotoxicological data to reach conclusions about both the nature and magnitude of injuries to specific natural resources. These differences are most pronounced in the development and use of toxicity reference values (TRVs).

Toxicity reference values are commonly used in ERA as toxicity thresholds for delineating acceptable and unacceptable risk and for translating protective levels to target media concentrations (i.e., clean-up levels or remedial goals). Toxicity reference values can be species specific or, more often, are developed for broad classes of receptors (e.g., benthic invertebrates, avian wildlife). The process of deriving TRVs typically entails a literature search and evaluation (sometimes of already compiled TRVs, such as ecological soil screening levels [EcoSSL], Sediment Quality Guidelines, Water Quality Criteria, Oak Ridge National Laboratory guidance) to identify the most applicable study or studies that will serve as the basis for the TRV (Sample et al. 1996). Ideally, response information would be translated from underlying studies with the use of a complete dose–response curve, which enhances understanding of changes in organism response with environmental contaminant levels. However, much toxicity literature is still reported in the form of no observable adverse effect levels (NOAELs) and lowest observable adverse effect levels (LOAELs).

The greater reliance on NOAEL/LOAEL data to derive TRVs, as opposed to quantitative effects concentrations derived from dose–response curves (e.g., effect concentration [EC_x] values), can be a significant barrier to coordination between ERA and NRDA. This greater reliance is associated with the intended use of the information (i.e., the derivation of “safe” vs “unsafe” environmental conditions). Exposure concentrations greater than TRVs only indicate situations in which injury or potential unacceptable risk could occur; however, information on the magnitude of response with increases in environmental concentrations is lost in this simplification. Adoption of alternative approaches would improve the use of toxicity information by incorporating the dose–response curve, such as the use of regression statistics to calculate effective concentrations to a certain percentage of test organisms (EC_x). Although increased transparency in this area would benefit both processes substantially, the magnitude information is particularly critical for the NRDA process, wherein response magnitude is necessary for injury quantification.

The use of NOAEL/LOAEL data to derive TRVs has other limitations and scientific drawbacks beyond their binary (threshold) nature (see, e.g., Stephan and Rodgers 1985; Hoekstra and Van Ewijk 1993; Chapman et al. 1996; Newman 2008). NOAEL and LOAEL values are largely artifacts of the design of the experimental dosing regimes and are not standardized across different studies to a specified magnitude of adverse effect. As a result, 2 studies could identify substantially different NOAELs/LOAELs; yet, the results of both studies may be pooled in calculating TRVs. For example, Moore and Caux (1997) compared regression approaches to pairwise hypothesis testing. The authors used 24 datasets that adequately fit at least 1 regression model and had at least 2 replicates per concentration. Hypothesis testing techniques applied to these same data produced NOAELs

that corresponded to EC_{xs} of between 10% and 40% effect. The LOAELs represented EC_x values of up to 76%. [Crane and Newman \(2000\)](#) also examined the correspondence between EC values and NOAELs. In 9 sets of round robin tests for fish growth effects, the median NOAEL value corresponded to an EC_x level of 10.5%. However, the ranges were large, with EC_x values between 3% and 34% effect. This level of uncertainty regarding degree of protectiveness and the magnitude of effect associated with exceedances of TRVs is rarely discussed in ERAs and greatly restricts their applicability in NRDA.

Effects assessment in both ERA and NRDA can rely heavily on ecotoxicological information to evaluate the effects of a stressor on the environment. Both processes utilize literature and site-specific data to define potential effects. However, limitations associated with the derivation and use of TRVs (as a single toxicity threshold) in ERA limits the ability to incorporate this ecotoxicological information into the injury assessment process. Both practices can benefit from greater use of more definitive effects assessment tools to capture the nature of dose–response relationships. When TRVs are derived in ERA, NRDA would benefit from more comprehensive presentation of the available underlying toxicological information in the effects assessment and improved documentation of the rationale for selection of critical studies. This may include the tabulation and evaluation of results from a broad range of published studies to facilitate analysis of concurrence, differences and trends across studies, test species (i.e., species sensitivity distributions), and test durations. Despite the acceptance of NOAEL/LOAEL data as a decision-making tool in ERA (likely because uncertainty associated with the use of NOAEL/LOAEL is manageable within the ERA decision process, e.g., the cost to support more precision might not be necessary to adequately support remedial decision making), adoption of alternative analytical approaches and greater transparency of the underlying ecotoxicological information would lead to greater coordination and consistency between ERA and NRDA.

Applicability of hazard quotients to characterize risk and injury

The standard approach for calculating and communicating ecological risk is deterministic hazard quotients (HQs). The HQ for each chemical–receptor combination is calculated by dividing the EEC or estimated dose to an organism by the TRV:

$$HQ = EEC/TRV \quad (1)$$

$$HQ = Dose/TRV \quad (2)$$

An HQ of 1 is used as the threshold for unacceptable ecological risk (USEPA 1997).

Despite the common use of the HQ approach in ERA, HQs have relatively little applicability to the NRDA process. This largely derives from 3 factors: 1) the spatial and temporal horizon of EECs adopted in ERA might not address the full range of NRD needs, 2) the TRV derivation procedures and associated threshold-based approaches to inferring toxicity might not be readily applicable to NRD data needs, and 3) translation of HQ to a meaningful quantification of injury might not be possible. Because the 1st 2 factors have been

discussed previously, we focus here on the limitations of the use of HQs to quantify injury.

Broadly speaking, HQs solely provide information on whether estimated exposure concentrations are below or greater than a derived response threshold. They provide no (or potentially misleading) information on the magnitude (i.e., quantification) of adverse effect ([Pastorok et al. 2002](#)). For example, $HQ = 10$ does not imply 2 times more effect (risk or injury) than $HQ = 5$ because the slopes of the underlying dose–response curves are not factored into the simple ratios. Similarly, $HQ = 2$ for 1 compound or species is unlikely to be associated with a similar magnitude of adverse effect for a different compound or species (again, because the underlying toxicological response data are neither normalized across compounds and species, nor are they generally retained in the analysis). Finally, the previous discussion on the broad range of adverse effects associated with NOAELs and LOAELs used to derive TRVs raise questions regarding the use of $HQ = 1$ as the threshold for significant adverse effects.

Natural resource damage assessment entails both the determination and quantification of natural resource injuries. As noted above, HQs do not provide quantitative information regarding the degree of anticipated adverse effects. As a result, HQs generally might not be sufficient to address both the injury determination and quantification phases in NRDA, and reliance on their use in ERA greatly limits the opportunity for coordination between ERA and NRDA.

Alternatives to deterministic HQs are available that would provide for greater consistency between ERAs and NRDA ([Sorensen et al. 2004](#)). For example, where appropriate data are available, probabilistic risk methods can be used to represent variability in exposure concentrations (EEC) and in ecotoxicological data (TRV), and dose–response regression models can be used in lieu of threshold values to derive TRVs ([MacIntosh et al. 1994](#)). Such approaches enable calculation of the distribution of expected adverse responses (or, alternatively, probabilities of exceeding specific adverse effects levels at different locations and times). Alternatively, the use of ranking schemes that categorize the concentration response of different stressors might provide for an enhanced degree of risk quantification ([Landis and Wiegers 1997](#)). Finally, it should be emphasized that HQs are not required in ERA. On the contrary, USEPA (1998) guidelines for ERA and others ([Menzie et al. 1996](#)) advocate use of multiple lines of evidence to evaluate each assessment endpoint. Lines of evidence, such as field- and laboratory-based approaches to assessing exposure and adverse effects at contaminated sites might be directly applicable to NRDA and might encourage greater coordination between ERA and NRDA.

CONCLUSIONS

Ecological risk assessment and NRDA consider the potential adverse effects of hazardous substance exposure on ecological resources and ecosystem processes and share a number of data inputs and analytical constructs. For example, both types of analysis generally entail development of an understanding of exposure to hazardous substances and consequent responses of environmental receptors to such exposures. However, the unique programmatic objectives of the 2 processes also give rise to divergent data requirements and analytical approaches. As a result, the development of a single, integrated assessment methodology is neither practical nor desirable.

Although institutional and programmatic impediments exist to integration of the 2 processes, opportunities to coordinate technical and scientific elements of the assessments are being capitalized on at a number of locations. Indeed, it is increasingly common to find some measure of integration or coordination between ERA and NRDA at contaminated sites. Although it is important to recognize that distinctions might exist in the spatial and temporal domains of the 2 analyses, as well as the nature of data needed to make decisions, opportunities for data sharing exist, particularly for the characterization of environmental exposures, as well as the derivation of ecotoxicological information for a number of response measures. In sum, effective coordination is not precluded by the underlying science. Rather, willing project participants, accommodating project schedules, and recognition of potential efficiencies associated with shared data collection can all lead to enhanced coordination and consistency between ERA and NRDA.

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