ORNL/TM-2003/17

A Framework for Net Environmental Benefit Analysis for Remediation or Restoration of Petroleum-Contaminated Sites

January 2003

R. A. Efroymson
J. P. Nicolette
G. W. Suter II

Environmental Sciences Division

ORNL/TM-2003/17

Environmental Sciences Division

A FRAMEWORK FOR NET ENVIRONMENTAL BENEFIT ANALYSIS FOR REMEDIATION OR RESTORATION OF PETROLEUM-CONTAMINATED SITES

Rebecca A. Efroymson
Environmental Sciences Division
Oak Ridge National Laboratory

Joseph P. Nicolette
CH2M Hill
Atlanta, GA

Glenn W. Suter II
National Center for Environmental Assessment
U.S. Environmental Protection Agency
Cincinnati, OH

January 2003

Prepared for
U.S. Department of Energy
National Petroleum Technology Office
Budget Activity Number AC 10 15 00 0

Prepared by
OAK RIDGE NATIONAL LABORATORY
Oak Ridge, Tennessee 37831
managed by
UT-BATTELLE, LLC
for the
U.S. DEPARTMENT OF ENERGY
under contract DE-AC05-00OR22725

CONTENTS

LIST OF FIGURES

Page

ACKNOWLEDGMENTS

Research sponsored by the Fossil Energy Office, U.S. Department of Energy, under contract DE-AC05-00OR22725 with UT-Battelle, LLC. The views in the paper do not necessarily reflect the policies of the U. S. Department of Energy or the U.S. Environmental Protection Agency. We thank F. Dexter Sutterfield, Nancy Comstock, Kathy Stirling, and Daniel Gurney, project managers at the DOE National Petroleum Technology Office in Tulsa, Oklahoma, for their support of this project. We thank Jim Loar of Oak Ridge National Laboratory, Randy Bruins of the U.S. Environmental Protection Agency, and three petroleum industry reviewers for comments on an earlier draft of this manuscript.

ABSTRACT

Efroymson, R. A., J. P. Nicolette, and G. W. Suter II. 2003. A framework for net environmental benefit analysis for remediation or restoration of petroleum-contaminated sites. ORNL/TM-2003/17. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Net environmental benefits are the gains in environmental services or other ecological properties attained by remediation or ecological restoration, minus the environmental injuries caused by those actions. A net environmental benefit analysis (NEBA) is a methodology for comparing and ranking the net environmental benefit associated with multiple management alternatives. A NEBA for chemically contaminated sites typically involves the comparison of the following management alternatives: (1) leaving contamination in place; (2) physically, chemically, or biologically remediating the site through traditional means; (3) improving ecological value through onsite and offsite restoration alternatives that do not directly focus on removal of chemical contamination or (4) a combination of those alternatives. NEBA involves activities that are common to remedial alternatives analysis for state regulations and the Comprehensive Environmental Response and Liability Act, response actions under the Oil Pollution Act; compensatory restoration actions under Natural Resource Damage Assessment, and proactive land management actions that do not occur in response to regulations: i.e., valuing ecological services or other ecological properties, assessing adverse impacts, and evaluating restoration options. This paper provides a framework for NEBA, with special application to petroleum spills in terrestrial and wetland environments. A high-level framework for NEBA is presented, with subframeworks for natural attenuation (the contaminated reference state), remediation, and ecological restoration alternatives. Primary information gaps related to NEBA include: non-monetary valuation methods, exposure-response models for all stressors, the temporal dynamics of ecological recovery, and optimal strategies for ecological restoration.

1. INTRODUCTION

Net Environmental Benefit Analysis (NEBA) is a methodology for identifying and comparing net environmental benefits of alternative management options, usually applied to contaminated sites. Net environmental benefits are the gains in environmental services or other ecological properties attained by remediation or ecological restoration[1], minus the environmental injuries caused by those actions. A NEBA for chemically contaminated sites typically involves the comparison of the following management alternatives: (1) leaving contamination in place; (2) removing the contaminants through traditional remediation; (3) improving ecological value through onsite or offsite restoration that does not involve removing contaminants; or (4) a combination of those alternatives. Examples of combinations include remediation of localized soil contamination combined with natural attenuation and the planting of trees, and the dredging of sediment hotspots combined with local wetland restoration. NEBA involves valuing ecological services or other properties, assessing adverse impacts, and evaluating restoration options. These activities are common to remedial alternatives analysis for state regulations and the Comprehensive Environmental Response and Liability Act (CERCLA); response actions under the Oil Pollution Act (OPA); compensatory restoration actions under Natural Resource Damage Assessment (NRDA), and proactive land management actions that do not occur in response to regulations. NEBA often incorporates the comparative methodology of habitat equivalency analysis (HEA), as described below. This methodology is in common use for ecological restoration alternatives related to petroleum spills. However, NEBA has not been formalized in a manner analogous to the U. S. Environmental Protection Agency (EPA) ecological risk assessment framework (EPA 1998), and land managers would benefit from such a framework for NEBA.

NEBA may be thought of as an elaboration of ecological risk assessment. That is, it is risk-benefit analysis applied to environmental management actions. Hence, the EPA ecological risk assessment framework (EPA 1998) could be adapted to perform NEBAs. However, because risk assessment does not normally consider benefits, and risk assessors are not familiar with the requirements of an assessment that estimates benefits, a new framework is useful to accentuate the specific features of such analyses. In addition, NEBAs are usually performed by resource management agencies that are not familiar with the ecological risk assessment formalism.

NEBA has the potential to help land managers avoid the possibility that the selected remedial or ecological restoration alternative will provide no net environmental benefit over natural attenuation of contaminants and ecological recovery. An alternative may provide no net environmental benefit because: (1) the remedial or ecological restoration action is ineffective (the action does not substantially change the risk) or (2) the remediation alternative causes environmental injuries greater than the damage associated with the contamination because (a) the need for remediation has been driven by human health risk, not ecological risk; (b) the ecological injury from contamination has been overestimated because of conservative assumptions; or (c) injuries associated with remediation were not properly addressed. Pitfall 2c is emphasized in this discussion. Similarly, NEBA has the potential to help land managers plan an ecological

---

[1]Restoration, as defined here, refers to actions that directly improve ecological services or other ecological properties, onsite or offsite (the term “mitigation” is sometimes used), in contrast to remediation, which focuses on chemical removal. Ecological restoration encompasses restoration, rehabilitation, replacement or acquisition of the equivalent, as defined by the Oil Pollution Act of 1990 (NOAA 1997).

restoration alternative that provides a positive net environmental benefit over the hypothetical state that would prevail in the absence of contamination. NEBA is recommended if any of the remedial or restoration alternatives potentially has significant negative ecological effects or minimal ecological benefits. Finally, NEBA is needed when the multiple alternatives are beneficial, but the one with the greatest net benefits is not apparent without formal analysis.

This paper provides a framework for NEBA; demonstrates how residual injuries and benefits from natural attenuation, traditional remediation, and ecological restoration options may be compared systematically; and identifies key research needs. Principal aspects of the framework include: (1) a single planning phase for analysis of all alternatives, (2) the identification of a comprehensive set of ecological services or other focal ecological properties, (3) the modular layout of certain components of the framework (e.g., chemical exposure and effects analysis), (4) the development of temporally variable estimates of exposure (e.g., due to biodegradation), (5) the development of credible, non-conservative exposure-response relationships beyond simple toxicity thresholds or habitat area thresholds, (6) the development and integration of temporal estimates of effects (e.g., due to recovery), and (7) the consideration of habitat equivalency and other potential valuation metrics for comparing ecological states. The emphasis of this paper and examples herein is on petroleum contamination in terrestrial and wetland ecosystems, although the framework is equally applicable to aquatic environments.

Environmental management alternatives must also be considered in an economic context, but cost issues (such as relative costs of alternatives, monetary value of ecological resources, costs of monitoring, and NRDA liability costs) are not currently included in the framework; this framework addresses net environmental benefits rather than net economic benefits. Similarly, human health risks are typically external to NEBA, but would contribute significantly to most management decisions about chemical contamination. If substantial human health risks are present, the relative net environmental benefit of alternatives would hold less weight in the decision.

2. PRECEDENTS FOR NEBA

Several precedents for NEBA exist, but they provide little specific, procedural or methodological guidance for the assessment of contaminated sites. These range from federal and state government examples to industry examples. The term NEBA is not commonly used in CERCLA remedial feasibility analysis or NRDA contexts. It was probably coined by agencies and industries evaluating options for marine oil spills, as in the report published by the U.S National Oceanic and Atmospheric Administration (NOAA) in 1990 entitled *Excavation and rock washing treatment technology: Net environmental benefit analysis*. In that study, representatives of Exxon, NOAA, and the State of Alaska evaluated a remedial option for the Alaskan shoreline affected by the Exxon Valdez oil spill "to determine if there were net environmental benefits from the excavation and washing of oiled sediments [below 15 cm depth, with heated seawater], and return of treated sediments to the excavated site over natural cleansing and the use of approved 1990 treatments," i.e., manual removal, spot washing, and bioremediation (NOAA 1990). Although the study provided an analysis of the potential adverse impacts associated with the proposed remediation technology, and estimated relative recovery periods, it did not provide a framework or propose metrics for comparison of injuries and benefits from the alternative methods. The term NEBA is commonly associated with assessments of oil spill dispersants in marine environments (Baker 2001, Fiocco and Lewis 1999, Lunel et al. 1997).

In another example of NEBA, the net environmental benefit of dredging part of an estuary was investigated (J. P. Nicolette, CH2M Hill, personal communication, October 11, 2001; Rubin et al. 2001). The approach was to adopt an estuary-wide sediment services strategy. Many ecological services from the contaminated sediments had been lost due to biochemical reduction within the anaerobic environment, which caused toxic levels of ammonia. However, sedimentation was shown to be occurring at rates that were expected to reduce the bioavailability of contaminants. Although natural attenuation was viewed as an attractive option because of its cost and efficacy, the regulatory agencies were concerned about potential injuries that could occur during the natural attenuation process. Therefore, a restoration action was proposed to deliver sediment services with certainty to offset the potentially lost sediment services. Additional applications of NEBA are listed in Table 1, though most applications of HEA, a subset of NEBA, are not publicly available because of their use in litigation proceedings (Milon and Dodge 2001), and thus many more NEBAs (especially terrestrial NEBAs) have been performed than those of which we are aware.

Table 1. Examples of NEBA

Example	Reference
Net environmental benefit (NEB) of excavation and rock washing treatment technology versus natural attenuation and approved treatments, Exxon Valdez oil spill	NOAA 1990
Quantification of wetland mitigation from petroleum pipeline construction	Nicolette et al. 2001
NEB of natural attenuation versus pump and treat technology versus air sparge/vapor extraction of volatile organic compounds in groundwater	J. Nicolette, CH2M Hill, confidential source
NEB of dredging versus not dredging an estuary, with quantification of restoration needed to offset uncertainty in risk assessment	Rubin et al. 2001
NEB of seagrass and mangrove restoration, following undisclosed disturbance at John's Island, Palm Beach County, Florida	S. Friant, Entrix, personal communication, May 23, 2002
NEB of the use of dispersant following the grounding of the *Sea Empress* in Great Britain	Lunel et al. 1997
Quantification of compensatory restoration of salt marsh vegetation on dredge material placed on a barrier island, given impacts from oil spill in Lake Barre, Louisiana	Penn and Tomasi 2002
Quantification of replacement habitat to compensate for coral reef injuries from vessel groundings	Milon and Dodge 2001

Although the NEBA terminology is not normally used in CERCLA remediation assessments, the concept is included in the guidance from the EPA Office of Emergency and Remedial Response (Luftig 1999). "Even though an ecological risk assessment may demonstrate that adverse ecological effects have occurred or are expected to occur, it may not be in the best interest of the overall environment to actively remediate the site. At some sites, particularly those that have rare or very sensitive habitats, removal or *in-situ* treatment of the contamination may cause more harm (often due to wide spread physical destruction of habitat) than leaving it in place. . . . The likelihood of the response alternatives to achieve success and the time frame for a biological community to fully recover should be considered in remedy selection. Although most receptors and habitats can recover from physical disturbances, risk managers should carefully weigh both the short- and long-term ecological effects of active remediation alternatives and passive alternatives when selecting a final response." Similarly, the Great Lakes Water Quality Board recommends that "prior to embarking on sediment remediation, [one should] have developed some quantifiable expectation of result (ecological benefit) and a program to follow the predicted recovery" (Zarull et al.1999).

Individual scientists have espoused NEBA-like concepts and methods (Principe 1995; Baker 1999). P. P. Principe of the EPA National Exposure Research Laboratory has used the term "Ecological Benefits Assessment" to refer to a procedure that could be used to assess changes in resource service flows that would result from different management or control alternatives at large spatial scales (Principe 1995). Principe (1995) emphasizes the importance of benefits assessment, and describes a general taxonomy of benefits, but does not provide methodological guidance.

Baker (1999) advocates the use of NEBA for evaluating oil spill clean-up alternatives. She describes elements of a process for NEBA, including (1) collection of environmental data, characterization of environmental services, and description of the remediation method; 2) review of spill case studies that are relevant to the proposed remedial method; (3) prediction of likely environmental outcomes; (4) comparison of advantages and disadvantages of remediation and natural attenuation; and (5) balancing of advantages and disadvantages to proposed alternatives to make a decision. These elements of NEBA have not been formalized in a framework.

At least three states endorse NEBA-related concepts or methodologies. That is, these states allow or advocate the comparison of environmental benefits in environmental management legislation. In addition, New Jersey, Massachusetts, Louisiana, Arkansas, Connecticut, Alaska, Indiana, California, Pennsylvania, and Delaware have supported NEBA-type strategies for evaluating remedial alternatives.

(1) The Texas Commission on Environmental Quality (TCEQ) (formerly the Texas Natural Resource Conservation Commission, TNRCC) recommends "Ecological Services Analysis" as an option for contaminated sites where chemical concentrations exceed ecologically protective concentration levels (PCLs) but not human health PCLs (TNRCC 2001). The potentially responsible party may propose compensatory ecological restoration after quantifying benefits and risks associated with alternative remedial actions or natural attenuation. HEA, described below, is one recommended comparative methodology, and others may be proposed to natural resource trustees. In addition, ecological services analysis and compensatory ecological restoration require approvals of the natural resource trustees for the state of Texas, obtained through the TCEQ.

(2) The State of Florida Department of Environmental Protection (DEP) may enter into a voluntary "ecosystem management agreement"with regulated entities and other government entities if the DEP determines that "implementation of such agreement meets all applicable standards and criteria so that there is a net ecosystem benefit to the subject ecosystem more favorable than operation under applicable rules" and "implementation of the agreement will result

in a reduction in overall risks to human health and the environment compared to activities conducted in the absence of the agreements" (State of Florida 2001). This "team-permitting" approach to environmental management was proposed by the business community and supported by the Florida DEP (Barnett 1999).

(3) Recent revisions to Washington State's Model Toxics Control Act include provisions for a "Disproportionate Cost Analysis" for the consideration of incremental benefits and costs in the selection of a remedial alternative. The comparison of benefits and costs may be quantitative or qualitative and need not be monetary. The analysis includes an evaluation of residual risks that are associated with each alternative, such as whether or not remedies that are protective of human health are also protective of ecological receptors (Washington State Department of Ecology 2001).

3. ALTERNATIVE ACTIONS

As stated above, alternative actions are divided into three principal categories: natural attenuation, traditional remediation, direct ecological restoration, and combinations of these. Comparisons are based on not only the state of contamination, but also ecological recovery.

3.1 NATURAL ATTENUATION

Natural attenuation is remediation through natural dilution and degradation processes, without addition of electron acceptors, nutrients, or electron donors. This alternative is equivalent to the "baseline" scenario for which risks are rather rigorously assessed in CERCLA remedial investigations in order to determine whether remediation is needed (EPA 1989, Sprenger and Charters 1997, Suter et al. 2000). Typically in CERCLA, the emphasis is on current risks and one or two future time points, rather than a continuous temporal analysis. If estimated health and ecological risks are sufficiently low, no remedial action is required and the contaminants are naturally attenuated. If these risks are unacceptable, natural attenuation is considered along with remedial alternatives that involve removal of contaminated media or interventions to increase the rate of attenuation. Natural attenuation may be chosen as the best alternative or part of the best alternative for meeting remedial goals if active remediation is ineffective, cost-prohibitive or damaging to the environment. Swindoll et al. (2000) provide a list of six situations where natural attenuation may be appropriate, including "there is no evidence of an imminent threat to ecological resources" and "sufficient time is available for [natural remediation]." Because active remediation may introduce new risks, natural attenuation may be a viable option even if Swindoll's two criteria are not met. One of EPA's criteria for judging natural attenuation is "to achieve site-specific remediation objectives within a time frame that is reasonable compared to that offered by other more active methods" (EPA 1999). If these remediation objectives relate to ecological properties, then net environmental benefit determinations should reflect estimates of ecological recovery. Natural attenuation is nonintrusive and has no incremental remedial hazards, only those associated with the original contamination and its metabolites. Few data are available to compare the risk reduction provided by natural attenuation to reductions from various remediation alternatives (Stahl and Swindoll 1999). Performance monitoring is important for this alternative (Heath 1999).

3.2 TRADITIONAL REMEDIATION

Excavation, incineration, burning, chemical remediation, microbial bioremediation, and phytoremediation reduce risks by removing contamination or actively reducing chemical concentrations in environmental media. Excavation is the most common option for remediating contaminated soils if the scale of contamination does not make the cost prohibitive. A physically harsh remedial alternative, such as soil excavation, would usually have greater, immediate adverse impacts to ecological receptors than concentrations of petroleum hydrocarbons at many spill sites, especially given that many semi-volatile hydrocarbons and their metabolites are not highly toxic to plants. Facilitated bioremediation can range from simple aeration (tilling) of soil to the addition of electron donors or microorganisms. Phytoremediation of petroleum hydrocarbons enhances rates of degradation in rhizosphere soil (Susarla et al. 2002). Some remedial alternatives, such as burning of spills in marshes and fields, are used only in emergency management situations (API 1999). Potential hazards posed by remedial interventions are listed in Table 2.

Rigorous assessments are not typically required to evaluate risks associated with remedial alternatives, and few guidance documents emphasize the importance of comparing risks from various remedial alternatives and no-action alternatives (Suter et al. 2000, Reagan 2000). Remediation is assumed to reduce risk. Remedial goals are defined based on health or ecological risks from the contaminants, but the remedial technologies are chosen based primarily on two engineering criteria: the ability to achieve those goals and cost-effectiveness. This focus on engineering criteria rather than environmental goals tends to restrict the range of options considered.

NEBA might have facilitated more rigorous ecological comparisons of alternatives to support past remedial actions. For example, at a Department of Energy facility in Oak Ridge, Tennessee, thousands of healthy, but PCB-contaminated, fish and other aquatic organisms were rotenoned to prevent a probable reproductive decrement to a few individual herons, ospreys and kingfishers feeding at the pond currently and in the future. Although substantial resources were devoted to assessing risk to the piscivorous birds, little effort was devoted to assessing the risk from the removal action. Similarly, recent research suggests that dredging of sediments in a canal of the San Francisco Bay may not have provided net environmental benefits, as measured by DDT and metabolite body burdens, and capping, more rigorous dredging, or an unevaluated ecological restoration alternative might have provided an environmental benefit (Weston et al. 2002).

3.3 ECOLOGICAL RESTORATION

Ecological restoration is the direct restoration of certain ecological entities (services or other properties of populations, communities, or ecosystems) or their habitats (specific wetland, grassland, forest, or stream bed types). In NRDAs, ecological restoration may be proposed by potentially responsible parties to replace time-integrated, lost services or other ecological properties, in lieu of monetary compensation. The restoration may occur on the affected land or on other land, usually in the same ecosystem. In either case, restoration is “compensatory,” damages are “offset,” and the net environmental benefit compared to the uncontaminated reference state is zero or positive. Ecological restoration of chemically contaminated land is

Table 2. Examples of ecological hazards posed by terrestrial remedial actions

Remedial Action	Hazard
Microbial Bioremediation and Phytoremediation	Possibly increased bioavailability or toxicity of hydrocarbons or products Devegetation due to tilling Decreased plant diversity and aqueous contamination due to fertilization
Excavation or Isolation (capping) of Soil	Destruction of vegetation Destruction of habitat and outmigration by vertebrates in excavated area Removal of nutrient-rich surface soil and associated microorganisms and invertebrates Failure of soil ecosystem and vegetation to recover if nonindigenous fill soil is used Destruction of ecosystem at borrow pit where fill is obtained and at landfill where excavated soil is deposited. Alarm and escape behavior of wildlife due to construction activity and noise
Burning of Spills, Soil Incineration or Thermal Desorption	Decrease in air quality and associated risk to wildlife or plants Destruction of above-ground vegetation, below-ground seeds and root material from severe heat Destruction of soil organic matter and potential loss of productivity Change in chemistry of oil residue which may prevent emergence of new shoots Secondary fires, extending area of habitat destruction Outmigration by vertebrates in burned area
Most Remedial Actions	Destruction of vegetation and outmigration by vertebrates in areas where roads, parking areas or laydown areas are developed, or foot traffic is frequent Reduction in biodiversity and wildlife forage from mowing of excavated area, cap or landfarm to maintain lawn Decrease in air quality associated with increased truck traffic

sometimes combined with a localized remedial action, such as hot spot removal, or with monitored natural attenuation (TNRCC 2001). Offsite ecological restoration could provide a net environmental benefit with lower costs than excavation or bioremediation of decades-old, refinery-contaminated land. Ecological restoration with natural attenuation would usually be expected to provide a net environmental benefit, compared to natural attenuation, because restoration provides benefits beyond the reduction in chemical risk through time.

Replacement habitats have included seagrasses, coral reefs, tidal wetlands, salmon streams, estuarine soft-bottom sediments, mangroves, mud flats, salt marshes, riparian forests, dune and swale ecosystems, and grasslands (http://contaminants.fws.gov/Issues/Restoration.cfm). For example, a 50-foot wide buffer zone of native trees and shrubs were planted on the eroded banks of a tributary of the Potomac River near Reston, Virginia, where a pipeline released diesel fuel overland and into the stream. The replacement habitat is sometimes located at a distance from the degraded habitat, particularly if site selection criteria are narrow. For example, seagrass restoration requires a specific substrate (Fonseca et al. 2000).

Phytoremediation of terrestrial oil spills may also restore services. Planting native plant species would restore primary production and wildlife habitats as well as aid in the removal of petroleum contamination, likely resulting in substantial net environmental benefit under NEBA. An example of the use of phytoremediation for this dual purpose is the planting of the marsh grass *Spartina alterniflora*, supplemented with fertilizer in a petroleum-contaminated wetland (Lin and Mendelssohn 1998).

Research is needed to define optimal strategies for ecological restoration of particular ecological services and other ecological properties. Habitat which appears to be successfully restored may support few individuals of a critical species or may not support sufficient reproduction to balance mortality, thereby becoming a sink habitat that drains individuals from other areas. A restored tidal marsh failed to create habitat for the endangered light-footed clapper rail (Hackney 2000). Restoration that involves physical construction may subsequently fail, resulting in ecological injuries (e.g., stream channel restorations that wash away or artificial wetlands that are dry). Indeed, ecological restoration technologies tend to be evaluated on the basis of engineering criteria, such as the ability to establish soil cover or to stabilize stream banks, rather than ecological criteria. Estimates of restoration endpoints may be uncertain due to temporal and spatial variability in precipitation and other environmental factors, natural variation in growth of vegetation and animals, errors in site preparation and in use of transplant material, predation on transplanted organisms, and human land use changes (Thom 2000, NRC 1992). In addition, droughts or floods may affect the success of restoration. NRDA consent decrees typically specify performance criteria and monitoring schedules (Penn and Tomasi 2002).

4. STRUCTURE OF NEBA FRAMEWORK

The high-level framework for NEBA is depicted in Fig. 1, and includes a planning phase, characterization of reference state, NEBA of alternatives (including characterizations of exposure of effects, including recovery), comparison of NEBA results, and possible characterization of additional alternatives. Only ecological aspects of alternatives are included in this framework. The figure also depicts the incorporation of cost considerations, the decision, and monitoring and efficacy assessment of the preferred alternative, although these processes are external to NEBA. Three subframeworks are presented: (1) characterization of the contaminated reference state (Fig. 2), (2) NEBA for a remediation alternative (Fig. 3), and (3) NEBA for an

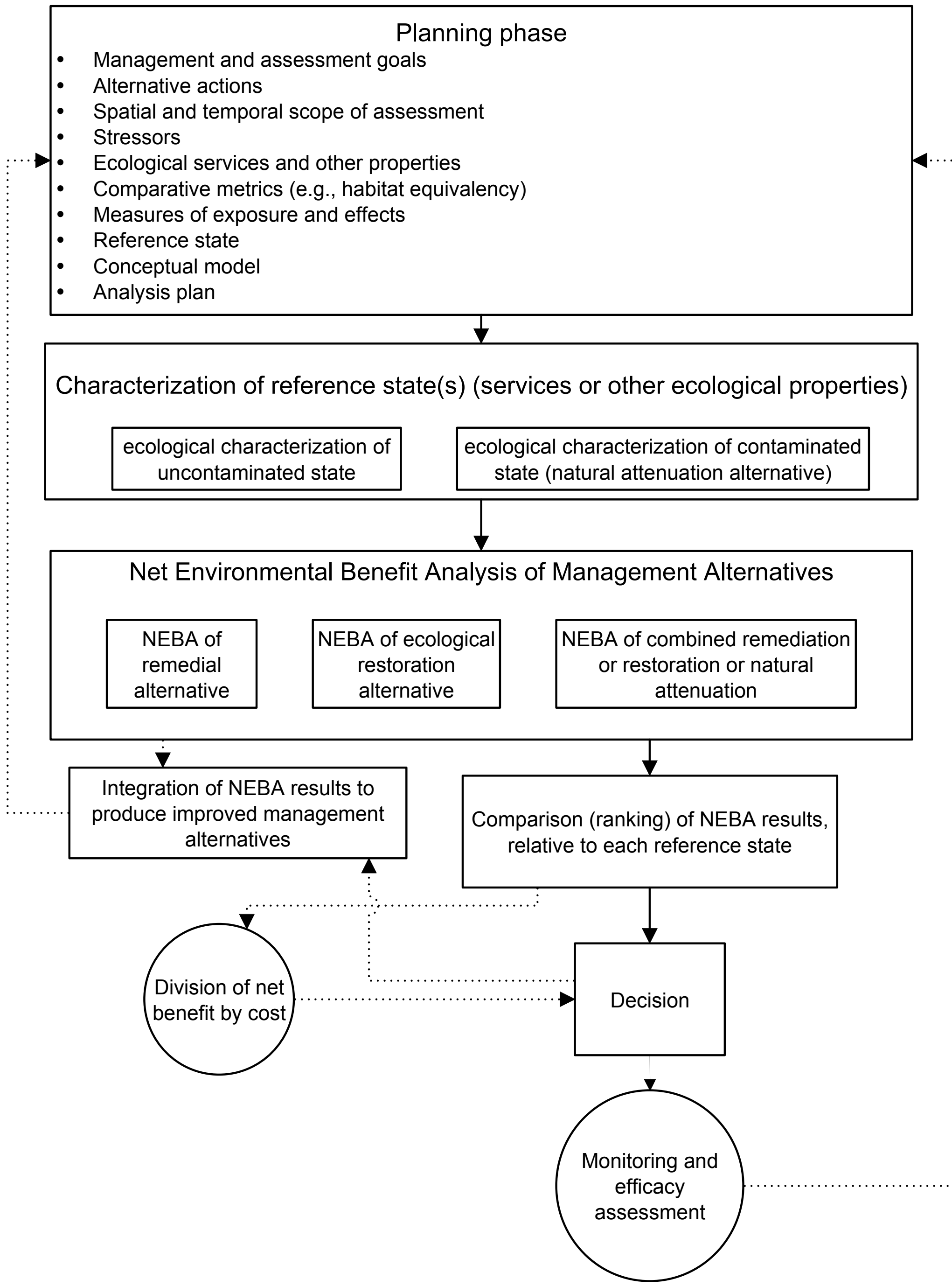

Fig. 1. Framework for Net Environmental Benefit Analysis. Dashed lines indicate optional processes; circles indicates processes outside of NEBA framework.

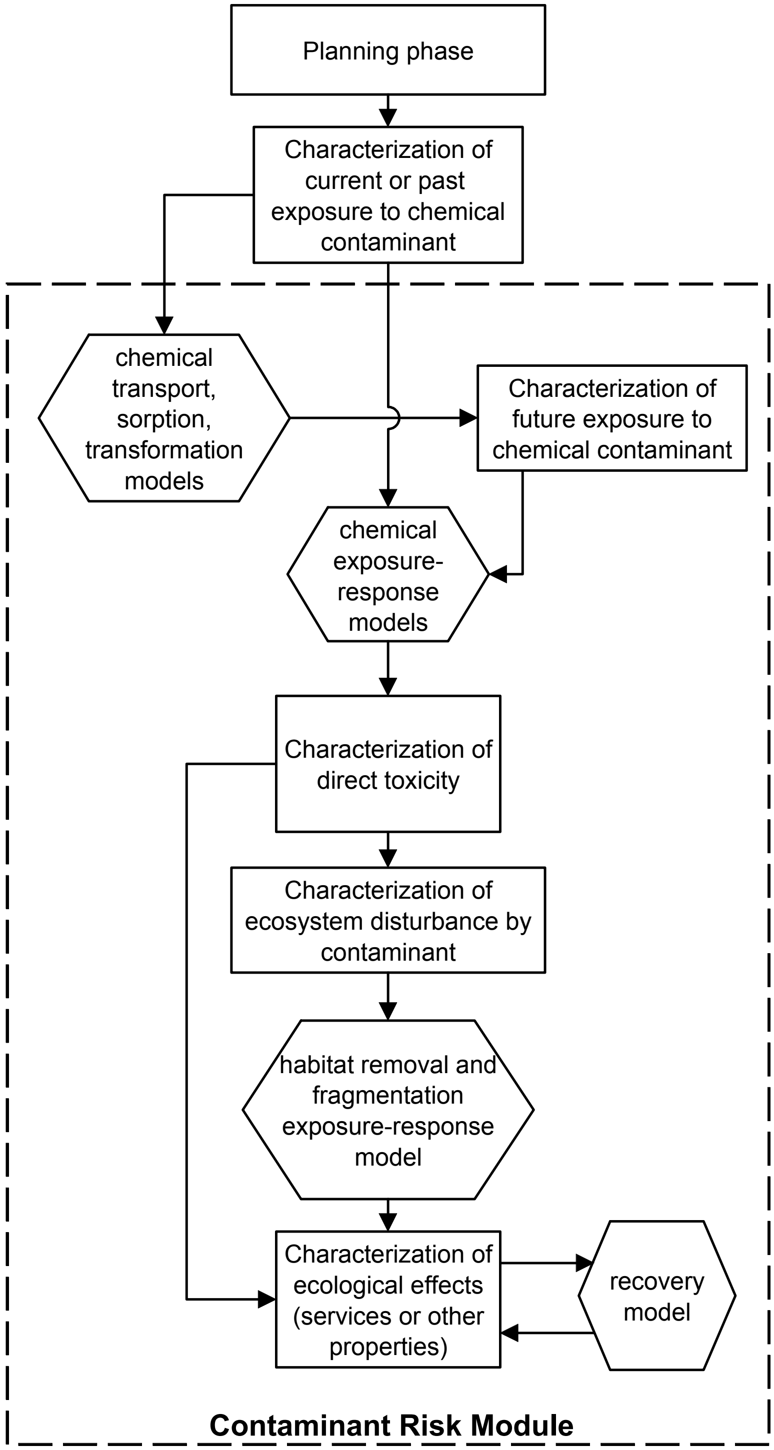

Fig. 2. Characterization of the contaminated reference state or natural attenuation. The net environmental benefit of natural attenuation, where the reference state is the trajectory of ecological entities (services and other properties) under contaminated conditions, is zero.

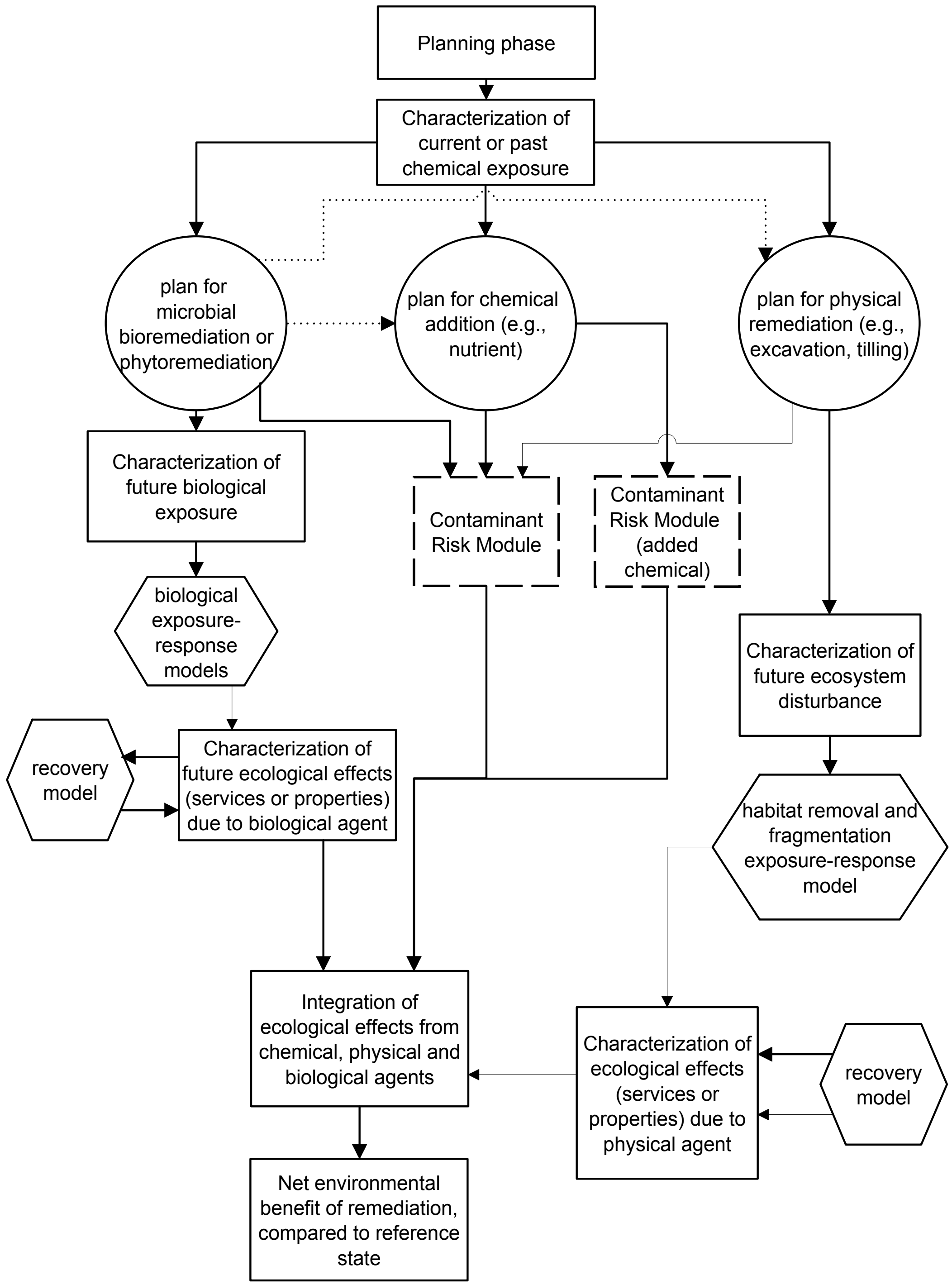

Fig. 3. Net Environmental Benefit Analysis of remedial alternatives. Dashed-borders on boxes indicate that the contaminant risk module in Fig. 2 should be inserted here.

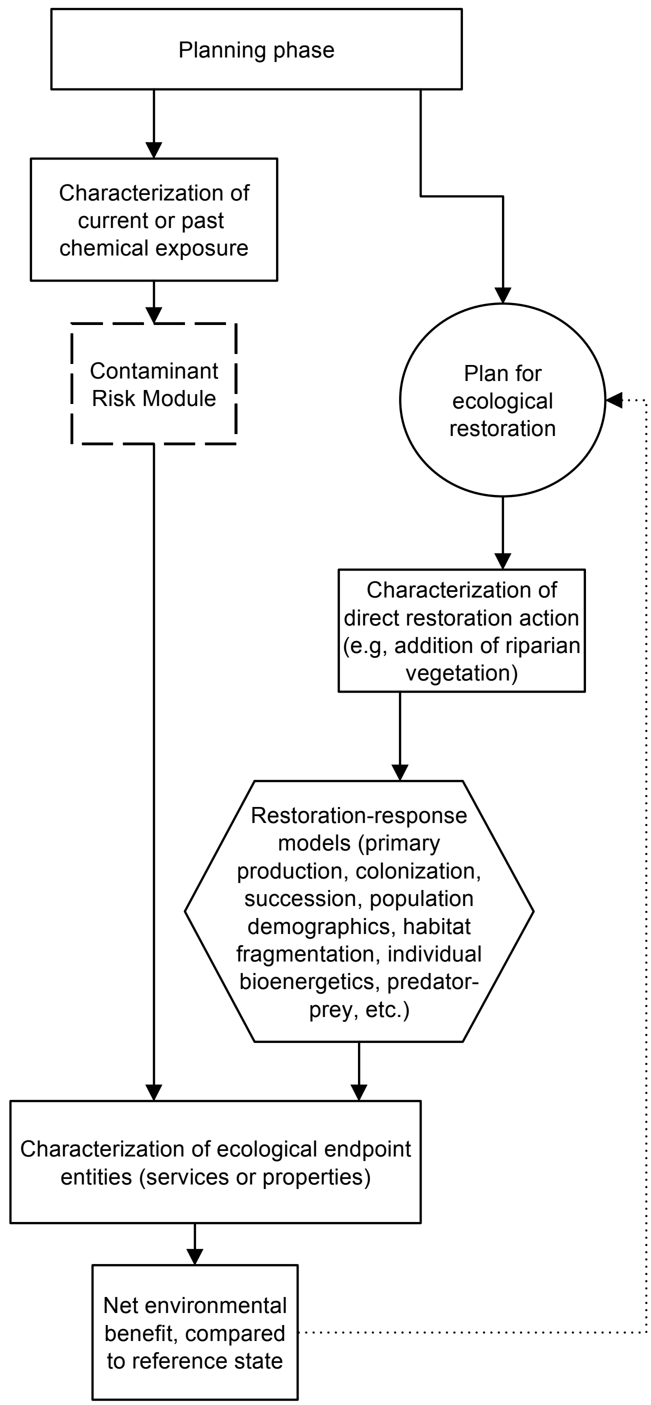

Fig. 4. Net Environmental Benefit Analysis of ecological restoration.

ecological restoration alternative (Fig. 4). Figure 2 also constitutes the steps of analysis for NEBA of natural attenuation, if natural attenuation is compared to the uncontaminated reference state. (The net environmental benefit of natural attenuation, compared to the contaminated reference state, is zero, by definition.) If an alternative involves multiple actions (e.g., addition of plants and chelation agents for phytoremediation or removal of hot spot contamination and grassland restoration), the assessor can draw on Suter (1999) for recommendations concerning how to estimate combined effects.

This detailed framework for NEBA does not preclude the use of more informal, NEBA-like approaches in regulatory negotiations. As in many ecological risk assessments, the funds available for for a NEBA may not allow the level of data collection that we recommend for estimating past, present and future ecological states with confidence.

4.1 PLANNING PHASE

The planning phase for a NEBA includes: setting the goals of assessment; selecting a limited and feasible suite of alternative actions (Sect. 3); defining the temporal and spatial scope of assessment; identifying contaminant and remediation stressors; selecting environmental services and other ecological properties of interest; selecting metrics and methodologies for the comparison of alternatives; selecting a reference state (Sect. 4.2); establishing a link between stressors and services (conceptual model); and developing an analysis plan (Fig. 1). The planning phase is comparable to the planning and problem formulation phases in a risk assessment (EPA 1998). A comparative assessment such as a NEBA should have a plan that encompasses all relevant, alternative actions. If NEBA is performed after the CERCLA or other baseline ecological risk assessment has been completed, the risk assessment and related data collection may need to be modified to suit the comparative purpose.

4.1.1 Management Goals

A common management goal for a NEBA may be to quantify net environmental benefits of remediation and restoration alternatives to support a cost-benefit analysis of those alternatives. Or, if a particular type of restoration is preferred by land managers and/or natural resource trustees, the management goal may be to restore land to the extent that there is a positive environmental benefit, compared to the uncontaminated reference state. For example, natural resources trustees selected marsh construction on dredge spoil as the preferred type of restoration at a pipeline spill site, and formal analysis focused on determining the amount of restoration needed to achieve the desired net benefit (Penn and Tomasi 2002). Rules and regulations, scoping assessments, or *ad hoc* decisions by regulatory agencies may define: (1) the ecological services or other properties of concern; (2) the relative importance of past, present and future injuries; (3) the reference state for the analysis (contaminated or uncontaminated); (4) acceptable or recommended analytical methodologies; (5) preferred comparative methods (HEA is recommended by TCEQ); or (6) preferred actions. For example, past damage is important in the NRDA context, because NRDA aims to compensate for ecological services lost in the past, present and future; but CERCLA remedial responses and related state regulations only draw on present and future conditions. One management goal may be to streamline the relationship between (1) risk assessment and response guidance of state regulations, CERCLA, and the Oil Pollution Act and (2) resource liability estimates associated with NRDA provisions of the federal acts. Emergency response may necessitate a decision before a formal NEBA can be undertaken.

4.1.2 Stressors

The stressor that is common to the determination of net environmental benefits of all alternatives is chemical contamination. Traditional remediation may impose the widest range of potential stressors, including the physical stressor of excavation or tilling, the biological stressor of introduced microorganisms or plants, the residual chemical stressor or added chelation agents, nitrate, or peroxide (Table 2). Nonchemical stressors are seldom considered under restoration scenarios, but it is possible that (1) vehicle movement, grading, tilling, or trampling could constitute stressors in the process of restoring an ecosystem, (2) the restoration may fail and result in physical damage, or (3) the restoration of habitat for one population could decrease habitat for another. For example, because killdeer prefer gravelly surfaces for laying their eggs, restoring soil and vegetation to these areas could reduce local populations. Similarly, the marsh restored on dredge spoil in Penn and Tomasi (2002) would have been of greater value to shorebirds if it had remained unvegetated. Moreover, the ecological service of protection from predation is in direct conflict with the service of provision of food to predators.

NEBA also considers benefits that result from a decrease in a chemical stressor or a direct ecological restoration effort. Although restoration is comprised of physical and biological components that could be termed "beneficial agents," we choose not to use that term because restoration feeds into the NEBA at the characterization of effects stage of analysis (Sect 4.3.3) and does not need to be separated into its component actions to determine exposure.

4.1.3 Ecological Services and Other Ecological Properties

NEBAs usually evaluate ecological services that are provided by an area of land or wetland. Services have been emphasized because (1) they are more easily valued than other ecological properties, prior to a cost-benefit analysis of alternatives; (2) services are the subject of the TCEQ's Ecological Services Analysis option (TNRCC 2001), and much petroleum activity is located in Texas; (3) ecological services are often the subject of NRDAs; and (4) HEA (discussed below) is a convenient methodology for comparing multiple services or multiple alternatives on a single scale.

The selection of services rather than population or community properties as focal entities of NEBA might appear to be inconsistent with CERCLA ecological risk assessments. Risk assessments associated with remedial investigations tend to emphasize multiple endpoint properties of organisms (e.g., mortality or fecundity) or populations (e.g., abundance or production) representing different trophic groups while NRDAs and NEBAs typically emphasize ecological services and ecosystem value. However, services estimated or measured in NEBAs are sometimes estimated or measured quantities in an ecological risk assessment (e.g., production of a plant community, abundance of a food item or area suitable for mating, nesting). In addition, the NEBA practitioner can choose other ecological properties as endpoints if they are consistent with the management goals of the assessment (e.g., regulations list injuries to survival, growth, reproduction, behavior, community composition, and community processes and functions as key components to NRDA (Department of Commerce 1996)). Barnthouse et al. (1995) note that resources and CERCLA assessment endpoints are "functionally equivalent," but the entities or properties may be different because trustees and CERCLA participants (DOE, EPA, state) emphasize different goals.

Environmental services or other ecological properties should be selected with all alternatives in mind. That is, if clapper rail habitat is injured or benefits from one alternative action, the state of clapper rail habitat under other alternatives, during the time and within the spatial extent of the analysis, should be obvious or evaluated.

NEBA should measure many ecological services or other ecological properties or demonstrate that the analysis of one is sufficient to represent others. The representation of all populations of a particular trophic level by a single species is often acceptable in CERCLA chemical risk assessments if species are not known to have differential sensitivity. Animals of similar taxa and feeding habits (e.g., insectivorous birds) are often assumed to have similar exposures and sensitivity to chemicals. However, if ecosystem area is lost during excavation or gained during restoration, the use of representative species would require that species home ranges and habitat requirements would be similar. More commonly, NEBAs use a single restoration metric to represent all services.

4.1.4 Comparative Metrics

Few comparative methodologies and metrics exist. The most common methodology for comparing ecological restoration alternatives at petroleum and other contaminated terrestrial and marsh sites is HEA. HEA "is a habitat[2]-based approach that determines compensation in terms of the amount of comparable habitat required to replace lost ecological services; [therefore], natural resource injuries must be determined at the habitat level" (DOI et al. 1999). A typical metric for comparing injured and replacement ecological services and other properties under HEA is the total service integrated over area and time, or service-hectare-year. The metric is often converted to present-day value.

In the simplest form of HEA, the analyst could assume that on a per area basis for a given plot of land, all ecological services are proportional to each other. That is, if grassland primary production is restored, then litter decomposition and the provision of nesting or lekking sites for all bird species will be restored, and the single metric of primary production is sufficient for the NEBA. In a more complex implementation of HEA, injuries that are not associated with ecosystem-level disturbance (e.g., direct mortality of birds from contact with oil spill, either measured or estimated) may be converted to habitat service metrics. Penn and Tomasi (2002) converted individual bird losses to the habitat area that would have produced the biomass, based on salt marsh production and inefficient energy exchange among trophic levels (but without explicitly considering potential nesting sites or habitat connectivity). Although habitat metrics simplify the comparative analyses, they are recommended for NEBA only if the correlations with all ecological services or other ecological properties are obvious or established in the NEBA. That is, a link should be made between the injured or restored ecosystem and the parameter used to represent the service flows from that ecosystem.

The major advantage of habitat equivalency metrics is that few analyses are performed. All ecological services are presumably represented by one or a few metrics, such as primary productivity.

However, at some sites habitat equivalency metrics could be improved. The result of a HEA or NEBA can be sensitive to the metric used to estimate the net environmental gains of ecological services or other ecological entities that are associated with a restoration alternative. For example, Strange et al. (2002) found that the marsh service metrics of (1) primary productivity, (2) provision of habitat for the endangered light-footed clapper rail (*Rallus*

---

[2]The term "habitat" in HEA refers to an ecosystem, rather than to species-specific habitat. In this paper, we attempt to use the term "ecosystem" to refer to land areas with ecological value based on their structure and functions and "habitat" to refer to species-specific habitat.

longirostris levipes), (3) provision of soil nitrogen, (4) provision of benthic invertebrate prey for fish and shellfish and (5) secondary productivity all resulted in different compensatory restoration quantities. In general, the "marginal contribution" of a land area to the abundance of an endangered species is not well understood (Unsworth and Bishop 1994). Thus, compensatory restoration should usually be determined based on multiple services, and the range of these results will provide an estimate of one source of uncertainty in the NEBA.

Similarly, HEA can be made more rigorous if the species for which habitat is assessed are specified. For example, rather than generic ecosystem equivalencies, species-specific habitat metrics, such as provision of suitable substrate for pitcher plant and sundew or provision of food for bog turtle could be evaluated. In that way, the quality of the habitat may be quantified in terms of the number of individuals supported or even the viability of the specified populations. Moreover, Milon and Dodge (2001) note that habitat equivalency is most applicable to uniform landscapes with little difference in biological functions across the injured area; thus they had to adjust basic habitat equivalency equations to account for different coral reef populations with different area uses and different recovery times.

In addition, it is recommended that practitioners of HEA consider modifications of simple habitat metrics that reflect more detailed habitat requirements of populations of concern, such as length of edge, connectivity of habitat, and minimum patch size required by the species. Total area of habitat is not a surrogate for the distribution of habitat. For example, if a hectare of land is damaged in the middle of a habitat corridor, the affected population is much larger than that which resides in the damaged area. Similarly, if replacement ecosystem is created at a distance from the injured ecosystem, the connectivity may be lost. The DOI considered wildlife forage range injury, which went beyond the damaged ecosystem, in its assessment of damages from the Colonial Pipeline Spill in Virginia (DOI et al. 1999). Moreover, the area of habitat lost is not correlated with population survival (and not a suitable metric for population injuries) when toxicity (not ecosystem area loss) reduces forage vegetation or prey, or bioaccumulation leads to toxicity in the ecological receptor.

Although some NEBA practitioners treat habitat equivalency service-area-year metrics as the principal, or even sole, metrics for NEBA, we believe that NEBA is a broader concept and that other comparative metrics and methodologies are worthy of discussion. For example, TCEQ (TNRCC 2001) will consider other comparative metrics for ecological services analysis. They state that "out-of-kind services can often be normalized such that they can be compared," though guidance on acceptable normalization methods is not provided. Habitat equivalency is an example of a *service-to-service* (or *resource-to-resource*) approach to scaling restoration actions (Chapman et al. 1998, NOAA 1995). Under the Oil Pollution Act, the preferred restoration actions are those that restore resources of the same type and quality and of comparable value as those injured. In contrast, in *valuation* approaches to scaling, lost and restored resources need not be of same type and quality. Values of the original and replacement resources are comparable[3] according to a chosen metric (Chapman et al. 1998, NOAA 1997).

As opposed to service-to-service approaches to comparing net benefits, valuation approaches to comparison require that equivalencies between different types of services or ecological properties be established (Chapman et al. 1998, NOAA 1997). In many cases, equivalencies are derived through regulatory negotiation; i.e., natural resource trustees may use economic valuation methods to establish adequate levels of compensatory services (NOAA 1995). Wetland compensation ratios are determined based on a combination of scientific criteria,

---

[3] or values equal the cost of the restoration plan, but this NRDA option is beyond the scope of this paper

negotiations among stakeholders, and the permit applicant's ability to pay (King and Adler 1991). The planning phase of a NEBA would have to reflect whether equivalent services or other ecological properties are those that are equally valued in economic terms by society, or whether ecological value is more important.

Numerous valuation methods are available to estimate and to compare apparent dollar values of ecological services[4]. The willingness to pay for some services can be inferred (termed a "revealed preference") from market prices or other estimates of present use of the resources. The cost of replacing ecosystem services may be an estimate of their value. To determine non-use values (also termed "intrinsic," "existence," or "passive" values) for ecological properties that are not traded in markets, the willingness to pay for an entity or the willingness to accept the loss of an entity can be expressed directly (contingent valuation, CV)[5] or derived from values of groups of attributes (conjoint analysis), both through surveys. The D.C. Circuit Court established through three cases that natural resource trustees are not limited to specific valuation methods, including CV (Jones 1997).

In a non-dollar approach to the comparison of ecological services or other properties, injuries of different types (human health and ecological risks) are classified as insignificant (*de minimus*), highly significant (*de manifestis*), or intermediate, and therefore requiring consideration of non-risk factors prior to a remediation decision (Suter et al. 1995). Implicit in this categorization is the assumption that an increased number of species, amount of area, or value of species (according to regulation or local preference) affected constitutes a greater injury. *De minimus* ecological risk, for example, is defined, based on regulatory precedents, as (1) "less than 20% reduction in the abundance or production of an endpoint population within suitable habitat within a unit area," (2) "loss of less than 20% of the species in an endpoint community in a unit area," or (3) "loss of less than 20% of the area of an endpoint community in a unit area." However, Suter et al. (1995) acknowledge that "the loss of all individuals from 20% of the range of a population can be considered equivalent to loss of 20% of individuals from the entire range of a population," except for the fact that these entities would be expected to recover at different rates. This type of comparative analysis is qualitative rather than quantitative, and would not apply to instances where restoration must offset damage exactly.

Another non-economic type of comparative valuation metric is the past available solar energy ("emergy") required to produce goods and services (Odum and Odum 2000). Thus, a deer population would have a higher value per joule than their food. This metric may be correlated with the recovery time for these entities following ecosystem removal (e.g., via excavation). In one example, the cost of constructing and operating Mississippi River diversions to marshes were compared with benefits using an emergy analysis, whereby "natural and human contributions required to construct and operate two diversions were expressed in common units of solar energy" (Martin 2002).

---

[4]Although this paper emphasizes ecological entities, environmental service valuation may encompass a wide range of human use values, such as the value of drinking and irrigation water.

[5] The precision of natural resource values is often low, as "willingness to accept" measures of value, which are appropriate for resource damage determinations (where something is lost), are often two or more times the analogous estimates of "willingness to pay" (Brown and Gregory 1999).

4.1.5 Temporal Measures of Exposure and Effects

NEBA analyzes ecological gains and losses associated with alternative management options through time. The time-scale of comparison should include the duration of injury combined with the longest construction and recovery period of the alternative actions, past, present and future. In NRDA, the lifetime of the replacement project is significant, if it is not expected to persist indefinitely (NOAA 1995). If future benefits are discounted, as in most NRDA analyses (Sect. 4.4.1), benefits after a few decades become negligible. Various stressors or restoration actions act continuously (e.g., persistent chemicals in environmental media), and others act almost instantaneously (e.g.,excavation). To estimate future effects, the rates of natural attenuation (biodegradation and aging through sorption to soil), the rates of contaminant removal or changes in bioavailability through remediation, and the rates of ecological recovery should be estimated. These dynamics may be incorporated in the characterization of exposure or the characterization of ecological effects (Fig. 2). Temporal analysis may be de-emphasized in a NEBA if (1) estimates of current ecological states are much more certain than estimates of future and past states and (2) temporal analysis is not required by the relevant statute.

4.1.6 Spatial Measures of Exposure and Effects

Because NEBA is a comparative analysis, it must include the largest spatial extent of analysis of any single alternative. That is, if ecological restoration is proposed one kilometer from the area of contamination, the state of the ecological services or other properties at that location must be ascertained under the competing, alternative scenarios. TCEQ (TNRCC 2001) requires that ecological restoration occur in the "same ecosystem." In addition to offsite restoration, offsite contamination is possible under natural attenuation and active remediation alternatives.

4.1.7 Conceptual Model

A conceptual model, a concept that is borrowed from risk assessment, is a graphical representation of the relationships between the chemical or nonchemical stressor and the responses of ecological services or other properties (Suter et al. 2000). If there is no connection between a stressor and a service, then the service does not need to be represented in the NEBA. Contaminant exposure pathways should be considered for all alternatives in the NEBA. For this reason, the "contaminant risk" module that is depicted in detail in Fig. 2 is also included in Fig. 3 and Fig. 4, the NEBAs for remediation and restoration, respectively. The conceptual model for NEBA of remediation alternatives should include stressor-service pathways for remedial technologies, such as the link between nutrients added in bioremediation and plant growth or diversity, or the link between excavation and vegetation cover (Fig. 3). If multiple alternatives include a particular remedial technology, such as dredging of hot spot contamination, this portion of the conceptual model should be depicted in all alternatives. If injuries of contaminants are indirect, i.e., if wildlife habitats or forage vegetation are directly affected by chemicals, but not the animals themselves, the connections between habitat or food and vertebrate population properties should be considered in the NEBA. Similarly, if the restoration plan calls for restoration of an ecosystem, the conceptual model should show how ecological services and other ecological properties are expected to be affected.

4.1.8 Analysis Plan

The analysis plan includes data collection, modeling, and logical analyses that are described or implicit in the NEBA framework. The plan should explain how exposure will be modeled, the exposure-response models that will be used, how recovery will be modeled, how net environmental benefits of different alternatives will be compared (e.g, habitat equivalency), and how uncertainty will be treated. Because a NEBA is a time-integrated analysis, the analysis plan should explain how predictions forward and backward in time will be made. Examples of assumptions include: first-order chemical degradation, instantaneous removal of plants during excavation, or linear recovery of an ecological property. The analysis plan should describe how sampling design decisions may influence the power to detect injuries relative to the reference state (Peterson et al. 2001). The analysis plan may describe NEBA results that would cause assessors to develop improved alternatives and to repeat the NEBA (Fig. 1).

4.2 CHARACTERIZATION OF REFERENCE STATE

NEBA involves the comparison of each alternative to a common reference state to determine net environmental benefit. Two potential reference states are (1) the contaminated reference state, equivalent to natural attenuation and ecological recovery, and (2) the uncontaminated reference state. Particular reference states may be mandated by regulations. For example, in NRDA, generally, a reference state consists of past, present and future conditions that would have prevailed in the absence of disturbance, i.e., the uncontaminated reference state (Fig. 1). In a CERCLA remedial investigation, the current state of the environment is typically characterized in the baseline assessment, and assessment endpoint properties associated with proposed remedies are compared to the contaminated reference state. The term “baseline” is avoided here because it has very specific but differing meanings in the CERCLA remedial investigation and OPA NRDA cases. If net environmental benefits of all alternatives are ranked, relative to the contaminated reference state, the ranking relative to the uncontaminated reference state should not differ, because the net environmental benefits in the two comparisons should differ from each other by the same constants. However, the absolute changes associated with each alternative depend on which reference state is chosen, and thus the choice of reference state for the NEBA could influence the decision.

Washington State’s Model Toxics Control Act requires that all remedial alternatives be compared to the most “practicable” permanent remedial option that is evaluated in the feasibility study (Washington State Department of Ecology 2001). If a NEBA ranking has been generated relative to the contaminated or uncontaminated reference state, the same environmental ranking should exist relative to the most “practicable” remedial option.

Characterizing the reference state is a challenge, and models may be required for analysis of a temporally changing reference. Ecological services and other properties that are associated with the uncontaminated state may sometimes be approximated by conditions at a neighboring, uncontaminated site or conditions prior to the disturbance. The U. S. Department of the Interior requires that injury quantification in NRDA be based on statistical comparisons between biological properties in assessment and uncontaminated reference areas (DOI 1995, Barnthouse and Stahl 2002). However, numerous environmental factors may have acted in concert with the contamination to alter ecological services following the chemical disturbance. Aquatic assessments typically utilize regional reference conditions that are bounded by analyses of several streams, but the adequacy of reference streams is always in question. To the extent possible, reference states are characterized by seasonal variability, meteorological variability, predator-prey cycles, and stressors that are not associated with the contamination or remediation or ecological restoration alternatives. However, in NEBAs, like in NRDAs, uncontaminated reference states are most often depicted as constant through time, because the variability is unknown (NOAA 1997).

The characterization of the contaminated reference state may precede or be a parallel component of a NEBA (Fig. 2). Current and future exposures are estimated, and exposure-response models are used to estimate injuries, as in CERCLA remedial investigations (Sect. 4.3.2 and Sect. 4.3.3). In Fig. 2, an indirect pathway whereby an ecosystem is disturbed and the areal disturbance results in injuries, is explicit. Oil and brine spills that occur at exploration and production sites often have little prolonged, direct toxicity, but large-scale ecosystem removal may result in injuries to populations. A recovery model is also explicitly included in the framework for characterization of the contaminated reference state (Fig. 2, Sect. 4.3.4).

4.3 NEBA OF SINGLE ALTERNATIVES

The net environmental benefit of each alternative is the benefit minus the injury of the alternative. If the net environmental benefit is positive (compared to the reference state), it is sometimes termed a credit; if it is negative, it is a debit. If the benefits are to different ecological services than the injuries, either both need to be expressed to the decision maker, or both should be normalized by a single metric. The comparative metrics discussed above apply.

The subframeworks for NEBA of remediation and ecological restoration are presented in Fig. 3 and Fig. 4, respectively. Remediation can introduce physical, chemical, or biological stressors that may partly or wholly balance the ecological benefits from reduced chemical concentrations. An ecological restoration alternative is designed to have beneficial effects on ecosystem-level ecological entities, but a rigorous NEBA is recommended. Both subframeworks include the estimation of exposure, the use of exposure-response models, and the estimation of recovery. In the ecological restoration subframework, the exposure analysis may be omitted, if the beneficial effects are well described in the restoration plan. However, the net environmental benefit of restoration should subtract the value of the services provided by the unrestored land.

The framework for NEBA of natural attenuation if the uncontaminated reference state is used is presented in Fig. 2. If the contaminated reference state is used in NEBA, the net environmental benefit of natural attenuation is zero, as these analyses are equivalent.

4.3.1 Time-integrated Analysis

The type of result that may be expected from a NEBA for each service or other ecological property is presented in Fig. 5. In this hypothetical example, following an oil spill the state of ecological services or other properties is rapidly degraded. However, the ecological property is expected to improve with time during natural attenuation of the contamination, followed by recovery (Fig. 5). The level of the ecological property associated with the uncontaminated reference state is assumed to continue at approximately the pre-spill level. Although natural variability is expected, the ecological property in the uncontaminated reference state is generally considered to be a constant (Fonseca et al. 2000). The proposed remedial alternative is expected to reduce the ecological property initially (as excavation would reduce vegetation production), but recovery is expected to be completed more rapidly than in the natural attenuation case (Fig. 5). In the proposed ecological restoration alternative, restoration is achieved more rapidly than

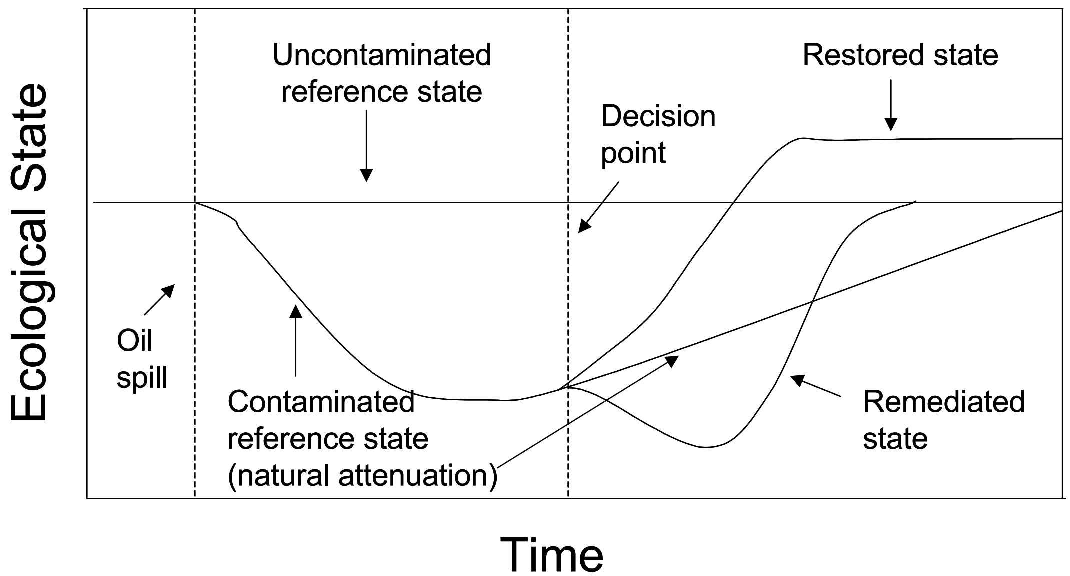

Fig. 5. Trajectory of assessment endpoint entity (service or other ecological property) with time, following a petroleum spill; conditions that would have been expected to prevail in the absence of the spill; expected trajectory of the remediated state; expected trajectory of the restored state.

ecological recovery in the natural attenuation alternative, and the final level of the ecological property is greater than the pre-spill level.

The net environmental benefit of remediation, compared to the contaminated reference state, is the area under the ecological property curve for the remediated state, minus the area under the ecological property curve for the contaminated reference state (Fig. 5). Note that in this instance, the net environmental benefit is close to zero and may be less than zero. The net environmental benefit of restoration, compared to the contaminated reference state (natural attenuation plus recovery), is above zero (Fig. 5). In one example of NEBA, the net benefit of planting to restore marsh services was calculated (Penn and Tomasi 2002). The spatial scope of the contaminated reference state included the unplanted marsh platform. In a non-monetary restoration under NRDA, restoration is intended to offset the loss in the past and prior to complete implementation of the alternative. Therefore, the net environmental benefit of ecological restoration, compared to the uncontaminated reference state, should be zero or greater to meet the NRDA management goal. The service is restored to a level above the pre-spill level in Fig. 5 to compensate for past loss. In practice, the target net environmental benefits of restoration might not be determined until after the injuries from chemical contamination are estimated, because restoration may be intended to offset exactly the injuries. Net environmental benefit is often expressed in integrated service-hectare-years. The type of analysis shown in Fig. 5 should be performed for each ecological service or other ecological property, or for a service (e.g., primary productivity) that represents many other services.

4.3.2 Characterization of Exposure

If benefits and injuries associated with alternatives are not obvious, they may be determined by exposure-response relationships if exposure is characterized. The characterization of exposure is the estimate of the magnitude of contact or co-occurrence of a stressor with an ecological service or other property. An analysis of a proposed ecological restoration alternative could omit the characterization of exposure (Fig. 4), unless excavation, construction, tilling, trampling or vehicle movement constitute significant stressors; benefits should be quantified in the restoration plan.

Present estimates of exposure to chemicals may be determined by the measurement of chemical concentrations in soil or water, with an assumption about the statistical distribution of unmeasured contamination. Past and future contamination can be estimated with simulation models or, if data permit, statistical forecasting or hindcasting. Although extensive and frequent biological surveys (with reference locations) may obviate the need for exposure analysis to estimate current effects of chemicals, these surveys would have to be accompanied by exposure measurements and non-conservative modeling to estimate future injury. Chemical exposures change through time through the processes of transport (leaching, volatilization), sorption, degradation, and transformation (Fig. 2). Changes in bioavailability should be estimated, as they are predictors of effects.

Exposures may be continuous or instantaneous. Chemical exposures are typically considered as continuous functions. However, if the change in exposure occurs over a time period that is short compared to the scale of the analysis (e.g, rapid degradation), such changes may be treated in a step-wise fashion. That is, the curves depicted in Fig. 5 would have vertical lines at certain time points if exposure was assumed to change instantaneously and ecological services were assumed to change instantaneously with exposure. The removal of chemicals by excavation may be assumed to cause instantaneous effects.

Because NEBA is a comparative analysis, exposure estimates for any particular stressor or alternative should not be expressed conservatively, though uncertainties may be noted. Conservative injury estimates for an alternative will lead to inappropriate ranking of alternatives in the comparative part of the NEBA. In typical chemical risk assessments, several conservative assumptions are made. An organism is often assumed to be exposed to the maximum, measured concentration of a chemical across space and time, or at the very least, an upper confidence limit on the mean of that concentration. Similarly, chemicals at non-detected concentrations are often assumed to be present at the chemical detection limit for the medium. Biodegradation is occasionally assumed to be zero. Disturbed ecosystems are sometimes assumed to be entirely unavailable to biota, even when they are partially utilized. None of these are valid assumptions for a comparative NEBA.

4.3.2.1 Biodegradation

Determining the rate and extent of biodegradation is relevant to NEBAs for natural attenuation and enhanced bioremediation. More models exist to aid in the estimation of biodegradation rates in groundwater than in surface soil to which ecological receptors are exposed. Rates depend on the concentrations of chemicals in soil, status of the microbial populations, and soil types, among other factors. During a field test of phytoremediation, a first-order model explained local chemical disappearance at some locations but not others (Nedunuri et al. 2000). The assumption of rapid, first-order kinetics in soil is often erroneous, because of the nutrient and oxygen limitations, insolubility of the bulk of a hydrocarbon mixture, sequestration of hydrophobic constituents in soil pores, potential toxicity of chemicals and byproducts, differential degradation of different hydrocarbon constituents, seasonal changes in rates, and time required for microbial acclimation (Odermatt 1997, Duncan et al. 1999, Samson et al. 1994). Dibble and Bartha (1979) have shown a good correlation between the rate of disappearance of hydrocarbons and monthly average temperatures in the field.

One researcher has developed a predictive method for estimating the average extent of petroleum hydrocarbon degradation in land farms, based on initial hydrocarbon composition (Huesemann 1995). Because total petroleum hydrocarbon concentrations appear to level off by 20 weeks of treatment in this study, it may be advisable in a NEBA to treat the change in hydrocarbon concentration as instantaneous, if the scale of a NEBA is two decades or more. If degradation effectively ceased, only the proportion of chemical degraded would be needed to determine exposure.

4.3.2.2 Bioavailability

Although changes in the bioavailability of hydrocarbons and other chemicals are known to occur (e.g., aging), the rates of the sorptive and diffusive processes that contribute to these changes are difficult to estimate. Ongoing research may provide rate constants for these processes, as well as bioavailability factors for ingestion of hydrocarbons by mammals and birds.

4.3.2.3 Chemical removal

The dynamics of source removal and associated changes in exposure may be adequately estimated in a remedial work plan. As stated above, excavation of soil to remove chemicals may be treated as an instantaneous process, with respect to chemical contamination.

4.3.3 Exposure-response Relationships

NEBA practitioners may determine the trajectory of ecological services or other properties through time, either directly, or based on one or more exposure-response models. However, continuous chemical exposure-response relationships are rarely available for species in soil. Exposure-response thresholds (Lowest Observed Adverse Effects Concentrations) or estimated EC50s (median effective concentrations) are commonly available for soil contamination, but the roles of soil type, receptor taxa, multiple chemicals, aging of chemicals, and acclimation to toxicity are not well understood. Moreover, the magnitude of the exceedence of a screening value does not reveal much about the magnitude of risk above the threshold (e.g., the percentage of the community that is injured) or the probability of injury, unless the exposure-response relationship is known. In some aquatic NRDAs, agencies have used exceedences of water and sediment-quality criteria as evidence of injury; however, elevated chemical concentrations are not, by themselves, reliable indicators of adverse natural resource effects (Barnthouse and Stahl, 2002).

In addition, toxicity threshold estimates tend to be conservative, and, as stated above, conservatism should play no role in a comparative NEBA. Ecotoxicological benchmarks or screening values, which usually represent estimates of thresholds, are conservative: they tend to represent low values in the distribution of toxic thresholds (e.g., 10^{th} percentile of values for phytotoxicity, Efroymson et al. 1997), and they are often based on tests in soils to which chemicals have been freshly added. Thus, toxicity of aged chemicals is sometimes not observed in the field at these concentrations (Suter et al. 2000). Tests of field soils or measurements of effects in the field should be relied on to the extent possible.

Future injuries from chemical contamination may be estimated for NEBA by field measurement or modeling to determine current ecological states, combined with (1) modeling of changes in exposure, followed by the use of toxicity and ecological relationships or (2) modeling of recovery, under the assumption that contamination is below toxic concentrations (Fig. 2). Toxicity tests performed at multiple times can indicate the approximate rate of reduction of toxicity with time. For example, toxicity of diesel fuel to *Tradescantia* in artificial soil was significantly removed by four weeks after planting (Green et al. 1996). Similarly, bioremediation treatment (tilling, fertilization and liming) of a fuel-spill-contaminated soil led to the removal of phytotoxicity after 20 weeks (Wang and Bartha 1990). Marwood et al. (1998) recommend a battery of toxicity tests to monitor the progress of bioremediation. If tests are carried out in the field, estimates of exposure may not be needed.

In addition, chemical contaminants such as petroleum can act by disturbing ecosystems (Fig. 2). Therefore, models that relate area and distribution of disturbed ecosystems to population sustainability would also be useful for estimating injuries from brine scars, some petroleum spills (Fig. 2), roads, and trampled areas, as well as injuries from physical remedial alternatives (Fig. 3). In contrast, assessing the direct impacts of excavation may only require the spatial and temporal dimensions within which all vegetation is gone; thus Fig. 3 shows no exposure-response model for physical remediation. Models that may be required to estimate ecological services and other properties under the restoration alternative may include processes of: primary production, colonization, succession, population demographics, bioenergetics, and predation (Fig. 4).

The confidence with which injuries should be demonstrated in a NEBA may depend on the regulatory context. Under NRDA, an injury must be demonstrated with a high degree of confidence, rather than with the "potential" qualifier that is sometimes used in risk assessment (DOI 1995).

4.3.4 Recovery

Ecological recovery is a key determinant of net environmental benefit but is difficult to quantify. Recovery typically refers to the colonization, growth or succession of an ecological entity, following the effective removal of the direct pressure of a stressor. Recovery defines the end of the NEBA analysis. As depicted in Fig. 2 and Fig. 3, recovery modeling estimates the reduction over time of the effects of contamination or of remedial actions. Recovery models may also apply to ecological restoration alternatives where services or other ecological properties are restored as a consequence of the direct restoration goal. Although the term "recovery" is not used in Fig. 4, the restoration-response models may be equivalent to recovery models, but the rate of recovery is increased by the restoration action. Guidance from the TCEQ (TNRCC 2001) notes that "estimates of recovery time may come from literature, site-specific information, or other affected property investigations," but this information is often difficult to obtain. In addition, certain services may not be measurable at the spatial scale of the action. For example, a restored riparian wetland was too small for investigators to measure recovery of small mammal populations (Wike et al. 2000).

Estimates of recovered ecological services or other properties require either (1) a specific function or (2) an approximate time to recovery and an assumption about the curve shape needed to get there. Example times to recovery of ecological properties from petroleum mixtures in various ecosystems (mostly in northern climates) are summarized in Table 3. Examples of times

Table 3. Select studies of ecological recovery from petroleum spills

Endpoint	Disturbance	Time to Recovery (yr)	Reference
non-phytotoxic concentration of hydrocarbons in soil	fuel spills in lysimeters, followed by tilling, fertilization and liming	0.4	Wang and Bartha 1990
crop productivity	kerosene contamination, followed by tilling, fertilization and liming	1	Dibble and Bartha 1979
vegetation production	JP-4 fuel-contaminated soil in 40 Air Force bioassays (reflecting field contamination incidents) with sorghum or pinto bean	2 to 4 (0.5 with unstated mitigation measures)	Lillie and Bartine 1990
establishment of vegetation	crude oil applied to saturation at depth of 4 ft, with tillage, in Stillwater, Oklahoma	2	McKay and Singleton 1974
colonization by forbs	experimental crude oil spill of $10 L/m^2$ at Masters Vig, Northeast Greenland	3	Holt 1987
majority of vegetation cover	experimental crude oil spill of $10 L/m^2$ at Masters Vig, Northeast Greenland, moist plots	5 to 8, estimate	Holt 1987
vegetation cover, diversity	experimental crude oil spill of $10 L/m^2$ at Masters Vig, Northeast Greenland, dry plots	several decades, estimate	Holt 1987
vegetation diversity	experimental crude oil spill of $10 L/m^2$ at Masters Vig, Northeast Greenland, dry plots	>8, estimate	Holt 1987
establishment of shrubs	crude oil spills in tundra communities of Mackenzie Mountains, Northwest Territories, Canada	20	Kershaw and Kershaw 1986
vegetation cover and species diversity	crude oil spills in tundra communities of Mackenzie Mountains, Northwest Territories, Canada	>35	Kershaw and Kershaw 1986
complete vegetation cover	crude oil applied to saturation at depth of 4 ft, with tillage, in Stillwater, Oklahoma	5	McKay and Singleton 1974
establishment of vegetation	crude oil applied to saturation at depth of 4 ft, with tillage, in Stillwater, Oklahoma	>5	McKay and Singleton 1974
total vegetation cover	experimental crude oil spill of 3273 L on simulated pipeline trench near Tulita, Northwest Territories, Canada	1	Seburn et al. 1996

Table 3. Select studies of ecological recovery from petroleum spills

cover of majority of plant species	experimental crude oil spill of 3273 L on simulated pipeline right-of-way near Tulita, Northwest Territories, Canada	>3	Seburn et al. 1996
majority of vegetation cover	experimental crude oil spill sprayed at 9.1 L/m^2 on Low Arctic tundra near Tuktoyaktuk, Northwest Territories, Canada	10 to 15, estimate	Freedman and Hutchinson 1976
vegetation cover (proportions of shrub, moss and lichen cover)	crude oil sprayed on soil surface at 18L/m^2 in black spruce taiga forest, interior Alaska	#20	Racine 1994
vegetation cover	crude oil spill from pipe at 100-250L/m^2 in black spruce taiga forest, interior Alaska; locations with oil remaining in subsurface only	#20	Racine 1994
vegetation cover	crude oil spill from pipe at 100-250L/m^2 in black spruce taiga forest, interior Alaska; locations with asphalt-like surface oil remaining	>20	Racine 1994
diversity of vegetation	crude oil applied at 11L/m^2 to plots in Port Harcourt, Nigeria	>0.8	Kinako 1981
productivity of vegetation	crude oil applied at 11L/m^2 to plots in Port Harcourt, Nigeria	>0.5	Kinako 1981
unspecified endpoint in mangrove forests	oil spills	10 to 20, estimate	Booth et al. 1991

to recovery of ecological properties from physical stressors and bioremediation are summarized in Table 4. One type of estimate of the minimum time to recovery could be provided by the average age of the lost vegetation (Vasek et al. 1975). Recovery of total vegetation cover from petroleum spills may often occur more rapidly than recovery from physical disturbance, although comparative tests of both types of disturbance in the same ecosystem have not been undertaken. Recovery from summer spills may be slower than recovery from winter spills (Freedman and Hutchinson 1976). To determine whether or not studies of ecological recovery from various stressors are relevant to recovery from oil spills or oil spill remediation alternatives, additional studies of oil spill recovery are needed. Time periods preceding recovery may sometimes be extrapolated from measurements taken at another site; however, most data on recovery relate to aquatic rather than terrestrial ecosystems (Niemi et al. 1990, Booth et al. 1991, NOAA 1990), and few published studies relate to recovery from remedial actions (Tamis and Udo de Haes 1995).

The recovery trajectory is often assumed to be linear (NOAA 1995). In reality, the dynamics of recovery may be complex, for example, encompassing recovery from multiple

Table 4. Select studies of ecological recovery from types of disturbance that may be associated with remediation alternatives

Endpoint	Disturbance	Time to Recovery (yr)	Reference
soil structure	Wheel-rutted and other compacted soils from tree harvesting by skidder in Northern Mississippi	8 to 12	Dickerson 1976
soil structure	compaction of soil in Wahmonie, a Mojave desert ghost town	>100, estimate	Webb and Wilshire 1980
soil bulk density	compaction of soil from timber harvesting by bulldozer or skidder in New South Wales, Australia	>5	Croke et al. 2001
crop root weights	clay loam mechanically compacted in plow furrow in southwestern Minnesota	>9	Blake et al. 1976
vegetation climax community structure	burn of oil spill in high marsh in Texas	7-8, estimate	Tunnell et al. 1997, API 1999
vegetation community structure	desert road, subsequently adandoned,	>87	Bolling and Walker 2000
vegetation community structure	pipeline construction in the southern Mojave desert (trenching, piling and refilling)	hundreds, estimate	Vasek et al. 1975
two earthworm populations	open-cast coal mining and reclamation (ploughing and reseeding)	3 to 15	Rushton 1986
climax community of earthworms	thermal cleaning of soil	100, estimate	Tamis and Udo de Haes (1995)
climax community of earthworms	bioremediation	<10, estimate	Tamis and Udo de Haes (1995)

processes, such as soil compaction, colonization and succession of vegetation. Vasek et al. (1975) suggest that the recovery of properties of scrub vegetation, such as composition and percentage ground cover, would occur with sigmoidal temporal dynamics. The temporal dynamics of recovery can be estimated by monitoring during the assessment period, but two data points are never sufficient to establish the shape of the curve.

Factors that influence the recovery of ecosystems from disturbance include: current state, disturbance severity and frequency, successional history, history of disturbance, preferred state, management of the disturbance, and random factors such as weather (Fisher and Woodmansee 1994). Recolonization time is dependent on the size of the site and the proximity to a recolonization source. Species that are characteristic of early successional communities recover relatively rapidly from disturbance to colonize disturbed areas, due to their high reproductive rates and rapid dispersal mechanisms (Booth et al. 1991). These factors are incorporated into recovery models.

Restoration measures may reduce the time to recovery in remediation or natural attenuation scenarios. It should be noted that the recovery of one ecological service or other property can be impeded by restoration of another; for example, the maintenance of caps requires that deeply rooted vegetation and burrowing mammals be kept off a site, inhibiting recovery of some potential endpoints (Suter et al. 1993).

Recovery following excavation and landfarming is likely to be of longer duration than toxic effects of low concentrations of petroleum hydrocarbons. So, while biodegradation may be estimated as an instantaneous process, recovery should not be. The error associated with the use of linear estimates of recovery will depend on the duration of the recovery, relative to the time scale of the NEBA.

4.4 COMPARISON OF MULTIPLE ALTERNATIVES

Following the net environmental benefit calculation for individual alternatives, the net environmental benefits of each alternative are compared (Fig. 1). As stated in the previous section, benefits and injuries of different types can only sometimes be normalized by a single metric in the NEBA for single alternatives. Similarly, the net environmental benefits of multiple alternatives may be ranked in the NEBA only if normalizing metrics are available (Sect. 4.1.4). If net environmental benefits of different alternatives are expressed in different units, the land managers or trustees may rank the alternatives subjectively. However, the ranking is likely to be more acceptable to stakeholders if relative values were established during the planning phase.

If alternatives are compared to the contaminated reference state, both remediation and restoration alternatives may have positive net environmental benefit. However, if alternatives are compared to the uncontaminated reference state, and the analysis includes the period of past damage as in NRDA, a contaminant removal alternative may ultimately provide the level of ecological services that were lost, but will never compensate for the past lost services and will not have a net environmental benefit. In NRDA compensatory actions, restoration is required to provide a net environmental benefit.

In a special case of NEBA, the net environmental benefits of each restoration alternative (relative to the uncontaminated reference state) are compared to the net injuries from contamination using HEA. As stated above, the management goal in NRDA and TCEQ's ecological services analysis is to compensate for the lost services. HEA is commonly used to identify ecosystem replacement projects that provide resources and ecological services of the types that have been lost and will be lost prior to the complete restoration. That is, the net environmental benefit, compared to the uncontaminated reference state, must be at least as great as the debit of services associated with contamination, compared to the uncontaminated reference state. HEA can include the monetary value of services, but only the ecological aspect is discussed here. HEA considers the recovery time-path of injured resources and services, the relative productivity of restored ecosystems, community succession in restored ecosystems, and the project life span (DOI et al. 1999). In one example of HEA related to petroleum damage, 7.5 ha of marsh plant strips were calculated to be needed for compensatory restoration, and 15.9 ha was the area more rapidly colonized because of the strips (Penn and Tomasi 2002).

If a single comparative metric is used in HEA (e.g., primary production service-hectare-years), a single graph of ecological services through time for each alternative (analogous to Fig. 5) can depict the dynamics from which relative net environmental benefits of alternatives can be calculated. If multiple services or multiple, species-specific habitats are explicitly considered, then multiple analyses of ecological services through time are performed, and compensatory restoration may be determined through a weight of evidence. The mathematics of HEA are illustrated in Penn and Tomasi (2002) and Milon and Dodge (2001). In addition to the temporal estimate of the service-hectare-years lost or gained, the discount rate is incorporated into the estimate.

4.4.1 Challenges of Comparative Assessments

The challenges of conducting comparative assessments within the NEBA framework include the few metrics available for relative valuation of alternatives or subtraction of risks from benefits within a single alternative (discussed above), as well as (1) incomparable conservatism among assessment results for individual alternative actions, (2) inadequate exposure-response models to quantify the absolute magnitude of risk, (3) limited utility of the weight-of-evidence approach to risk assessment, (4) relative valuation of differing magnitudes of uncertainty when comparing net benefits of competing alternatives, and (5) relative valuation of past, present, and future conditions within an alternative.

First, comparisons of ecological properties under remediation, natural attenuation, and restoration alternatives require assumptions of comparable conservatism. Ecological risk assessments commonly generate conservative estimates of exposure and effects, so estimates of injury generated independently of the NEBA analysis may be high (therefore a comprehensive planning phase for NEBA is suggested). (See discussions of conservatism in Sect. 4.3.2 and Sect. 4.3.3.)

Second, assessors may be limited by the uncertainty of existing, empirical models and measurements. We are much more confident in predicting the decrement in biomass of plants where the surface soil has been excavated (i.e., no vegetation) than we are in predicting the percentage biomass decrement where a particular concentration of petroleum hydrocarbons or metals is found.

Third, the weight-of-evidence approach that is common in ecological risk assessment seldom results in a single value for magnitude of injury, because it is usually intended to aid an assessor in determining whether or not injury is above reference levels. It is difficult to compare net benefits across alternatives (e.g., how much to increase productivity in a restoration scenario for that alternative to be recommended) when our most quantitative risk assessments are not very quantitative. A weight-of-evidence approach could support NEBA if multiple models or measurements resulted in different estimates of magnitude of injury or benefit. For example, a weighting of different metrics could be used to estimate compensatory restoration, as discussed above.

Fourth, uncertainty has a role in comparative ecological valuation. For example, Arrow and Fisher (1974) assert that uncertainty and irreversibility of ecological states can "lead to a reduction in net benefits from an activity with environmental costs." Irreversibility reduces the "option value," which is an important component of an environmental benefit calculation (Chavas 2000). However, uncertainty may make a NEBA comparison indeterminate if the uncertainty is much greater than the differences in net benefits among the alternatives.

Fifth, to compare present effects in one scenario to future effects in another, future effects may be discounted to present value, and past service flows should be compounded, through methods analogous to economic discounting. Discounting is typically practiced in HEA, as well as in non-HEA NEBAs. NOAA usually recommends applying a three percent rate for discounting interim losses and gains (NOAA 1997), but the concept of discounting needs to be more thoroughly examined by practitioners and managers of NEBA before a recommendation is made. Some investigators recommend a zero discounting rate for ecological functions with no economically quantifiable human use (Milon and Dodge 2001). Moreover, NEBA results are highly sensitive to the choice of discounting rate (Milon and Dodge 2001). The discount rate should be adjusted if the value of services is not constant over time, for example, if the marginal value of wetland increases because its land area is decreasing, or if the marginal value of wetland decreases because the cost of creating new wetlands or substituting for its services is reduced (Unsworth and Bishop 1994).

5. ADDITIONAL CONSIDERATIONS

NEBA provides important information about environmental benefits and injuries of alternative actions to decision-makers. As shown in Fig. 1, cost-effectiveness is also an important criterion for the decision, although it is outside of the NEBA framework. Essentially, the net environmental benefit, divided by cost, results in an estimate of cost-effectiveness. Human health risk will probably also inform the decision, and sometimes a human health risk assessment or screening analysis may be required for a NEBA to be considered by regulatory agencies (TNRCC 2001). The OPA regulations specify criteria for selecting a preferred restoration alternative: (1) cost of the alternative; (2) "extent to which each alternative is expected to meet the trustees' goals and objectives in returning the injured natural resources and services to baseline and/or compensating for interim losses" (NOAA 1997); (3) likelihood of success; (4) extent to which the alternative will prevent future injury from the incident and avoid collateral injury from implementing the alternative; (5) extent to which the alternative benefits more than one natural resource or service; and (6) effect of the alternative on health and safety (NOAA 1997). All of these criteria are compatible with NEBA, and #2 and #4 recommend NEBA-type analyses.

A NEBA may be performed iteratively as alternative actions are optimized, preferably before an action is implemented (Fig. 1). Monitoring and efficacy assessment may be considered external to the NEBA framework, but these processes produce data that may result in a decision to alter the preferred alternative and possibly to perform the NEBA again. The Great Lakes Water Quality Board recommends "that much greater emphasis be placed on post-project monitoring of effectiveness of sediment remediation" (Zarull et al. 1999). Restoration actions sometimes fail to meet their goals, and adaptive management can lead to effective redesign of restoration alternatives (Thom 2000), especially if ecological properties have been well-specified in the NEBA planning phase. The NEBA framework is open in that it does not demand that net environmental benefit be the only criterion for a decision-maker.

6. CONCLUSIONS

A framework for Net Environmental Benefit Analysis (NEBA) of contaminated sites has been developed. NEBA is expected to be useful for decision-making related to petroleum spills and other contaminated sites. NEBA takes a holistic approach to decision-making for chemical contamination, considering both expected risks from remedial actions and direct benefits that are possible through ecological restoration. The methodology is expected to be most useful for decisions where certain types of injuries (e.g., from remedial actions) or benefits (e.g., from direct ecological restoration or enhanced remediation) are not consistently, rigorously assessed. NEBA provides a methodology whereby state and federal remedial investigations and NRDAs may be undertaken under a single framework. Similarly, NEBA provides a framework whereby proactive, nonregulatory-driven restoration may occur. A comparative assessment such as a NEBA should have a single planning phase that encompasses all relevant, alternative actions: natural attenuation, traditional remediation, and restoration alternative(s). Comparative metrics should be agreed upon. NEBA utilizes a reference state (contaminated or uncontaminated) that is consistent with management goals. A NEBA may result in additional alternatives that are optimized for net environmental benefit or that are targeted to provide offsetting environmental benefit, compared to the uncontaminated reference state. Supporting knowledge that requires further development includes: non-monetary valuation metrics; non-conservative, quantitative exposure-response models; models of recovery; and strategies for ecological restoration. The use of NEBA should result in better decisions, resulting in greater improvements in environmental quality at lower cost.

7. REFERENCES

API (American Petroleum Institute). 1999. Compilation and review of data on the environmental effects of *in situ* burning of inland and upland oil spills. Health and Environmental Sciences Department. Publication Number 3684. Washington, D.C.

Arrow, K. J., and A. C. Fisher. 1974. Environmental preservation, uncertainty, and irreversibility. *Quarterly J. Econ.* 88:312–319.

Baker, J. M. 1999. Ecological effectiveness of oil spill countermeasures: How clean is clean? *Pure Appl. Chem.* 71:135–159.

Barnett, E. 1999. Florida's incentive-based environmental protection: Alternative regulatory approach. Presentation. Environmental policies for the new millennium: Incentive- based approaches to environmental stewardship. Conference, November 2-3 1999. World Resources Institute. Washington, D.C. http://www.igc.apc.org/wri/incentives/barnett.htm

Barnthouse, L. W., J. J. Bascietto, S. A. Deppen, R. W. Dunford, D. E. Gray, and F. E. Sharples. 1995. Natural Resource Damage Assessment Implementation Project: Savannah River Site. U.S. Department of Energy, Office of Environmental Policy & Assistance, Washington, D.C.

Barnthouse, L. W., and R. G. Stahl. 2002. Quantifying natural resource injuries and ecological service reductions: challenges and opportunities. *Environ. Manage*. 30:1–12.

Blake, G. R., W. W. Nelson, and R. R. Allmaras. 1976. Persistence of subsoil compaction in a mollisoil. *Soil Sci. Soc. Am. J.* 40:943–948.

Bolling, J. D., and L. R. Walker. 2000. Plant and soil recovery along a series of abandoned desert roads. *J. Arid Environ.* 46:1–24.

Booth, P. N., D. S. Becker, R. A. Pastorok, J. R. Sampson, and W. J. Graham. 1991. Evaluation of restoration alternatives for natural resources injured by oil spills. Publication Number 304. American Petroleum Institute. Washington, D.C.

Brown, T. C., and R. Gregory. 1999. Why the WTA-WTP disparity matters. *Ecolog. Econ*. 28:323–335.

Chapman, D., N. Iadanza, and T. Penn. 1998. Calculating resource compensation: an application of the service-to-service approach to the Blackbird Mine hazardous waste site. National Oceanic and Atmospheric Administration Damage Assessment and Restoration program Technical Paper 97-1.

Chavas, J.-P. 2000. Ecosystem valuation under uncertainty and irreversibility. *Ecosystems* 3:11–15.

Croke, J., P. Hairsine, and P. Fogarty. 2001. Soil recovery from track construction and harvesting changes in surface infiltration, erosion and delivery rates with time. *Forest Ecol. Manage.* 143:3–12.

Department of Commerce. 1996. Oil Pollution Act Regulations. Part 990. Natural Resource Damage Assessments. 15 CFR Part 990. http://www.epa.gov/superfund/programs/nrd/15cfr990.pdf

Dibble, J. T., and R. Bartha. 1979. Rehabilitation of oil-inundated agricultural land: a case history. *Soil Sci*. 128:56–60.

Dickerson, B. P. 1976. Soil compaction after tree-length skidding in Northern Mississippi. *Soil Sci. Soc. Am. J*. 40:965–966.

DOI (U.S. Department of the Interior). 1995. 43 CFR Subtitle A, Part 11. Natural Resource Damage Assessments.

DOI (U. S. Department of the Interior), Virginia Department of Environmental Quality, and District of Columbia Department of Health. 1999. Colonial Pipeline Oil Spill, Reston, Virginia. Final Restoration Plan and Environmental Assessment. National Park Service, U.S. Department of the Interior, Washington, D.C.

Duncan, K. E., R. Kolhatkar, G. Subramaniam, R. Harasimhan, E. Jennings, S. Hettenbach, A. Brown, C. McComas, W. Potter, and K. Sublette. 1999. Microbial dynamics in oil-impacted prairie soil. *Appl. Biochem. Biotechnol*. 77–79:421–434.

Efroymson, R. A., M. E. Will, G. W. Suter II and A. C. Wooten. 1997. *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants: 1997 Revision*, ES/ER/TM-85/R3. Oak Ridge National Laboratory, Oak Ridge, Tennessee.

EPA 1989. Risk Assessment Guidance for Superfund. Vol. I. Human Health Evaluation Manual. EPA 540/1-89/002. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency, Washington, D.C.

EPA 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, D.C.

EPA 1999. Use of Monitored Natural Attenuation at Superfund, RCRA Corrective Action, and Underground Storage Tank Sites. OSWER Directive 9200.4-17P. Washington, D.C.

Fisher, S. G., and R. Woodmansee. 1994. Issue paper on ecological recovery. pp. 7–1 to 7–54. in Ecological Risk Assessment Issue Papers. EPA/630/R-94/009. U.S. Environmental Protection Agency, Washington, D.C.

Fiocco, R. J., and A. Lewis. 1999. Oil spill dispersants. *Pure Appl. Chem.* 71:27–42.

Fonseca, M. S., B. E. Julius, and W. J. Kenworthy. 2000. Integrating biology and economics in seagrass restoration: How much is enough and why? *Ecol. Eng.* 15:227–237.

Green B. T, C. T. Wiberg, J. L. Woodruff, E. W. Miller, V. L. Poage, D. M. Childress, J. A. Feulner, S. A. Prosch, J. A. Runkel, R. L. Wanderscheid, M. D. Wierma, X. Yang, H. T. Choe, and S. D. Mercurio. 1996. Phytotoxicity observed in *Tradescantia* correlates with diesel fuel contamination in soil. *Environ. Exper. Bot.* 36:313–321.

Freedman, W., and T. C. Hutchinson. 1976. Physical and biological effects of experimental crude oil spills on Low Arctic tundra in the vicinity of Tuktoyaktuk, N.W.T., Canada. *Can. J. Bot*. 54:2219–2230.

Hackney, C. T. 2000. Restoration of coastal habitats: expectation and reality. *Ecolog. Eng.* 15:165–170

Heath, J. S. 1999. Introduction: guidance on natural attenuation in soils and groundwater. *J. Soil Contam*. 8:3–7.

Holt, S. 1987. The effects of crude and diesel oil spills on plant communities at Mesters Vig, Northeast Greenland. *Arctic Alpine Res.* 19:490–497.

Huesemann, M. H. 1995. Predictive model for estimating the extent of petroleum hydrocarbon biodegradation in contaminated soils. *Environ. Sci. Technol*. 29:7–18.

Jones, C. A. 1997. Use of non-market valuation methods in the courtroom: recent affirmative precedents in Natural Resource Damage Assessments. Water Resources Update, Issue 109, Autumn 1997. Universities Council on Water Resources, Carbondale, Ill. http//131.230.132.19500/pdf/V109_A3.pdf

Kershaw, G. P., and L. J. Kershaw. 1986. Ecological characteristics of 35-year-old crude-oil spills in tundra plant communities of the Mackenzie Mountains, N.W.T. *Can. J. Bot.* 64:2935–2947.

Kinako, P. D. S. 1981. Short-term effects of oil pollution on species numbers and productivity of a simple terrestrial ecosystem. *Environ. Pollut. Ser. A* 26:87–91.

King, D. M., and K. J. Adler. 1991. Scientifically defensible compensation ratios for wetland mitigation. U. S. Environmental Protection Agency Office of Policy, Planning and Evaluation. Washington, D.C.

Lillie, T. H., and R. W. Bartine. 1990. Protocol for evaluating soil contaminated with fuel or herbicide. pp. 198–203. In W. Wang W, J. W. Gorsuch, and W. R. Lower (eds.), *Plants for Toxicity Assessment*. ASTM, Philadelphia, Pa.

Lin, Q., and I. Mendelssohn. 1998. The combined effects of phytoremediation and biostimulation in enhancing habitat restoration and oil degradation of petroleum contaminated wetlands. Ecol. Eng. 10:263–274.

Luftig, S.D. 1999. Issuance of Final Guidance: Ecological Risk Assessment and Risk Management Principles for Superfund Sites. OSWER Directive 9285.7-28. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency; Washington, D.C.

Lunel, T., J. Rusin, N. Bailey, C. Halliwell, and L. Davies. 1997. The net environmental benefit of a successful dispersant operation at the *Sea Empress* incident. Pp. 185–194. In Proceedings of the 1997 International Oil Spill Conference. Fort Lauderdale, Fla.

Martin, J. F. 2002. Emergy valuation of diversions of river water to marshes in the Mississippi River Delta. *Ecolog. Eng.* 18:265–286.

Marwood, T.M., K. Knoke, K. Yau, H. Lee, J. T. Trevors, A. Suchorski-Tremblay, C. A. Flemming, V. Hodge, D. L. Liu, and A. G. Seech. 1998. Comparison of toxicity detected by five bioassays during bioremediation of diesel fuel-spiked soils. *Environ. Toxicol. Water. Qual.* 13:117–126.

McKay, D., and P. C. Singleton. 1974. Time required to reclaim land contaminated with crude oil–an approximation. Agricultural Extension Service, University of Wyoming, Laramie, Wyo.

Milon, J. W., and R. E. Dodge. 2001. Applying habitat equivalency analysis for coral reef damage asssessment and restoration. *Bull. Mar. Sci*. 69:975–988.

Nedunuri, K. V., R. S. Govindaraju, M. K. Banks, A. P. Schwab, and Z. Chen. 2000. Evaluation of phytoremediation for field-scale degradation of total petroleum hydrocarbons. *J. Environ. Eng*. 126: 483–490.

Nicolette, J. P., M. Rockel, and M. J. Kealy. September 2001. Quantifying ecological changes helps determine best mitigation. *Pipe Line Gas Ind.* 52–57.

Niemi, G. J., P. DeVore, N. Detenbeck, D. Taylor, A. Lima, J. Pastor, J. D. Yount, and R. J. Naiman. 1990. Overview of case studies on recovery of aquatic ecosystems from disturbance. *Environ. Manage*. 14:571–587.

NOAA Damage Assessment and Restoration Program. 1995. Habitat equivalency analysis: an overview. National Oceanic and Atmospheric Administration. Silver Spring, Md.

NOAA Damage Assessment and Restoration Program. 1997. Scaling compensatory restoration actions, guidance document for Natural Resource Damage Assessment. National Oceanic and

Atmospheric Administration. Silver Spring, Md.

NOAA Hazardous Materials Response Branch. 1990. Excavation and rock washing treatment technology: Net environmental benefit analysis. National Oceanic and Atmospheric Administration. Seattle, Wash.

NRC (National Research Council). 1992. Restoration of aquatic ecosystems: science, technology, and public policy. National Academy Press, Washington, D.C.

Odermatt, J. R.. 1997. Simulations of intrinsic biodegradation using a non-linear modification of first-order reaction kinetics. *J. Soil Contam.* 6:495–508.

Odum, H. T., and E. P. Odum. 2000. The energetic basis for valuation of ecosystem services. *Ecosystems* 3:21–23.

Penn, T., and T. Tomasi. 2002. Calculating resource restoration for an oil discharge in Lake Barre, Louisiana. *Environ. Manage*. 29:691–702.

Peterson, C. H., L. L. McDonald, R. H. Green, and W. P. Erickson. 2001. Sampling design begets conclusions: the statistical basis for detection of injury to and recovery of shoreline communities after the 'Exxon Valdez' oil spill. *Mar. Ecol. Prog. Ser.* 210:255–283.

Principe, P. P. 1995. Ecological benefits assessment: A policy-oriented alternative to regional ecological risk assessment. *Hum. Ecol. Risk Assess*. 1:423–435.

Racine, C. H. 1994. Long-term recovery of vegetation on two experimental crude oil spills in interior Alaska black spruce taiga. *Can. J. Bot.* 72:1171–1177.

Reagan, D. P. 2000. Natural remediation in the risk management process: goals, options, and monitoring, pp. 9–29. In M. Swindoll, R. G. Stahl, Jr., and S. J. Ells (eds.), *Natural Remediation of Environmental Contaminants: Its Role in Ecological Risk Assessment and Risk Management.* SETAC Press, Pensacola, Fla.

Rubin, R. R., M. J. Brennan, M. C. Bracken, R. L. Hirsch, C. Leonard, and F. M. Rundlett. 2001. Options for enhancing the transfer of excess and surplus military property for conservation or recreational use. Bahr, Inc. Middleton, Wis.

Rushton, S. P. 1986. Development of earthworm populations on pasture land reclaimed from open–cast coal mining. *Pedobiologia* 29:27–32.

Samson, R., C. W. Greer, T. Jawkes, R. Desrochers, C. H. Nelson, and M. St-Cyr. 1994. Monitoring an aboveground bioreactor at a petroleum refinery site using radiorespirometry and gene probes: effects of winter conditions and clayey soil. pp. 329–333. In R. E. Hinchee, B. E. Alleman, R. E. Hoeppel, and R. N. Miller (eds.), *Hydrocarbon Bioremediation*. Lewis Publishers, Boca Raton, Fla.

Seburn, D. C., G. P. Kershaw, and L. J. Kershaw. 1996. Vegetation response to a subsurface crude oil spill on a subarctic right-of-way, Tulita (Fort Norman), Northwest Territories, Canada. *Arctic* 49:321–327.

Sprenger, M. D., and D. W. Charters. 1997. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessment, Interim Final. Environmental Response Team, U.S. Environmental Protection Agency, Edison, N.J.

Stahl, R. G., and C. M. Swindoll. 1999. The role of natural remediation in ecological risk assessment. *Hum. Ecol. Risk Assess.* 5:219-223

State of Florida. 2001. Ecosystem Management Agreements. The 2001 Florida Statutes, Title XXIX Public Health, Chapter 403, Sect. 403.0752. http://www.leg.state.fl.us/Statutes/

Strange, E., H. Galbraith, S. Bickel, D. Mills, D. Beltman, and J. Lipton. 2002. Determining ecological equivalence in service-to-service scaling of salt marsh restoration. *Environ. Manage*. 29:290–300.

Susarla, S., V. F. Medina, and S. C. McCutcheon. 2002. Phytoremediation: an ecological solution to organic chemical contamination. *Ecol. Eng.* 18:647–658.

Suter, G. W., II. 1999. A framework for assessment of ecological risks from multiple activities. *Hum. Ecolog. Risk Assess*. 5:397–413.

Suter, G. W., II, B. W. Cornaby, C. T. Hadden, R. N. Hull, M. Stack, and F. Zafran. 1995. An approach for balancing health and ecological risks at hazardous waste sites. *Risk Analysis* 15:221–231.

Suter, G. W., II, R. A. Efroymson, B. E. Sample, and D. S. Jones. 2000. Ecological risk assessment for contaminated sites. Lewis Publishers, Boca Raton,FL.

Suter, G. W., II, R. J. Luxmoore, and E. D. Smith. 1993. Compacted soil barriers at abandoned landfill sites are likely to fail in the long term. *J. Environ. Qual.* 22:217–226.

Swindoll, M., R. G. Stahl, Jr., and S. J. Ells. 2000. Natural remediation of environmental contaminants: its role in ecological risk assessment and risk management, pp. 1–8 *In* Swindoll, M., R. G. Stahl, Jr., and S. J. Ells (eds.) *Natural Remediation of Environmental Contaminants: Its Role in Ecological Risk Assessment and Risk Management.* SETAC Press, Pensacola, FL.

Tamis, W. L. M., and H. A. Udo de Haes. 1995. Recovery of earthworm communities (Lumbricidae) in some thermally and biologically cleaned soils. *Pedobiologia* 39:351–369.

Thom, R. M. 2000. Adaptive management of coastal ecosystem restoration projects. *Ecol. Eng.* 15:365–372.

TNRCC 2001. Guidance for conducting ecological risk assessments at remediation sites in Texas. Toxicology and Risk Assessment Section, Texas Natural Resource Conservation Commission, Austin, Tex.

Tunnell, J. W., K. Withers, and B. Hardegree. 1997. Environmental impact and recovery of the Exxon pipeline spill and burn site, Upper Copano Bay Texas: Final Report. TAMU-CC-9703-CCS. Texas A&M University-Corpus Christi, Tex.

Unsworth, R. E., and R. C. Bishop. 1994. Assessing natural resource damages using environmental annuities. *Ecol. Econ.* 11:35–41.

Vasek, F. C., H. B. Johnson, and D. H. Eslinger. 1975. Effects of pipeline construction on creosote bush scrub vegetation of the Mojave Desert. *MadroÁo* 23:1–13.

Wang X, and R. Bartha. 1990. Effects of bioremediation on residues, activity and toxicity in soil contaminated by fuel spills. *Soil Biol. Biochem*. 22:501–505.

Washington State Department of Ecology. Amended February 12, 2001. Model Toxics Control Act. Chapter 173–340 WAC. Publication No. 94-06 Seattle, Wash. http://www.ecy.wa.gov/pubs/9406.pdf

Webb, R. H., and H. G. Wilshire. 1980. Recovery of soils and vegetation in a Mojave desert ghost town, Nevada, U.S.A. *J. Arid. Environ.* 3:291–303.

Weston, D. P., W. M. Jarman, G. Cabana, C. E. Bacon, and L. A. Jacobson. 2002. An evaluation of the success of dredging as remediation at a DDT-contaminated site in San Francisco Bay, California, USA. *Environ. Toxicol. Chem.* 21:2216–2224.

Wike, L. D., F. D. Martin, H. G. Hanlin, and L. S. Paddock. 2000. Small mammal populations in a restored stream corridor. *Ecol. Eng.* 15:S121–S129.

Zarull, M. A., J. H. Hartig, and L. Maynard. 1999. Ecological benefits of contaminated sediment remediation in the Great Lakes Basin. 1999. Sediment Priority Action Committee, Great Lakes Water Quality Board, International Joint Commission, Washington, D.C. http://www.ijc.org/boards/wqb/ecolsed/execsum.html

INTERNAL DISTRIBUTION

1. S. G. Hildebrand, 1505, MS-6037
2-6. R. A. Efroymson, 1505, MS 6036
7. D. S. Jones, 1505, MS 6036
8. M. J. Peterson, 1505, MS 6036
9-11. ESD Library
12. ORNL Central Research Library

EXTERNAL DISTRIBUTION

13. N. Comstock, U. S. Department of Energy, National Petroleum Technology Office, One West Third St., Suite 1400, Tulsa, OK 74103

14. J. P. Nicolette, CH2M Hill, 115 Perimeter Center Place, NE, Suite 700, Atlanta, GA 30346-1278

15. G. W. Suter II, National Center for Environmental Assessment, U. S. Environmental Protection Agency, MS117, 26 W. Martin Luther King Drive, Cincinnati, OH 45268