Primary Restoration

Guidance Document for Natural Resource Damage Assessment Under the Oil Pollution Act of 1990

Damage Assessment
and Restoration Program

August 1996

PRIMARY RESTORATION

GUIDANCE DOCUMENT FOR NATURAL RESOURCE DAMAGE ASSESSMENT UNDER THE OIL POLLUTION ACT OF 1990

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DISCLAIMER

This guidance document is intended to be used in the assessment of restoration of injured natural resources under the Oil Pollution Act of 1990 (OPA). This document is not regulatory in nature. Trustees are not required to use this document in order to receive a rebuttable presumption for natural resource damage assessments under OPA.

NOAA would appreciate any suggestions on how this document could be made more practical and useful. Readers are encouraged to send comments and recommendations to:

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PREFACE

The goal of the Oil Pollution Act of 1990 (OPA) is to make the environment and public whole for injuries to natural resources and natural resource services resulting from an incident involving a discharge or substantial threat of a discharge of oil (incident). This goal is achieved through returning injured natural resources and services to baseline and compensating for interim losses of such natural resources and services through the restoration, rehabilitation, replacement or acquisition of equivalent natural resources and/or services.

The NRDA regualtions supporting OPA provide a framework for conducting sound natural resource damage assessments that achieve restoration under OPA. This document focuses on the procedures that may be used to restore or replace natural resources injured as a result of an oil spill incident. The information contained in this document may best be used in conjunction with injury assessment and restoration plan development.

LIST OF ACRONYMS

ACOE, USACOE	United States Army Corps of Engineers
ADF&G	Alaska Department of Fish and Game
API	American Petroleum Institute
CDFG	California Department of Fish and Game
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act of 1980
EPA, USEPA	United States Environmental Protection Agency
EVOS	*Exxon Valdez* Oil Spill
EVOS-RPWG	*Exxon Valdez* Oil Spill, Restoration Planning Work Group
FDA	United States Federal Drug Administration
F&WS, USF&WS	United States Fish and Wildlife Service
DOC, USDOC	United States Department of Commerce
DOI, USDOI	United States Department of the Interior
DOT	United States Department of Transportation
HEP	Habitat Evaluation Procedure
HMRAD	Hazardous Materials Response and Assessment Division,
NOAA	National Oceanic and Atmospheric Administration
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service (NOAA)
NOAA	National Oceanic and Atmospheric Administration
NRDA	Natural Resource Damage Assessment
OPA	Oil Pollution Act of 1990
PRP	Potentially Responsible Party
RP	Responsible Party
USACOE, ACOE	United States Army Corps of Engineers
USCG	United States Coast Guard
USDA	United States Department of Agriculture
USEPA, EPA	United States Environmental Protection Agency
USF&WS, F&WS, USFWS	United States Fish and Wildlife Service
USDOC, DOC	United States Department of Commerce
USDOI, DOI	United States Department of the Interior
WES	United States Army Engineer Waterways Experiment Station

LIST OF ACRONYMS (continued)

Legend

ha	hectare(s)
m	meter(s)
hr	hour(s)
spp	species

INTRODUCTION CHAPTER 1

1.1 Background

A major goal of the Oil Pollution Act of 1990 (OPA)[1] is to make the environment and public whole for injury to or loss of natural resources and services as a result of a discharge or substantial threat of a discharge of oil (referred to as an *incident*). This goal is achieved through returning injured natural resources and services to the condition they would have been in if the incident had not occurred (otherwise referred to as *baseline* conditions), and compensating for interim losses from the date of the incident until recovery of such natural resources and services through the restoration, rehabilitation, replacement, or acquisition of equivalent natural resources and/or services.

The U.S. Department of Commerce, acting through the National Oceanic and Atmospheric Administration (NOAA), issued final regulations providing an approach that public officials (trustees) may use when conducting Natural Resource Damage Assessments (NRDA) under OPA.[2] These NRDA regulations (the OPA regulations) describe a process by which trustees may:

- Identify injuries to natural resources and services resulting from an incident;

- Provide for the return of injured natural resources and services to baseline conditions and compensation for interim lost services; and

- Encourage and facilitate public involvement in the restoration process.

The OPA regulations are included in Appendix A of this document for reference. The preamble discussion of the OPA regulations, along with a summary of and response to public comments received on the proposed regulations, is published at 61 Fed. Reg. 440 (January 5, 1996).

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[1] 33 U.S.C. §§ 2701 *et seq.*

[2] The OPA regulations are codified at 15 CFR part 990 and became effective February 5, 1996.

1.2 Purpose and Scope of this Document

The purpose of the Restoration Guidance Document is to review the state of the art for restoration of certain habitats and biological natural resources and evaluate potential restoration actions following injury to natural resources resulting from the discharge of oil. Trustees should refer to Appendix B for a listing of this and other related guidance documents in support of the OPA regulations.

The following tasks were conducted in developing this document:

- Identify and evaluate oil-related restoration methods/techniques that are currently available for feasibility, effectiveness and success, and costs. This evaluation is performed on each habitat and biological natural resource (species population) of concern in aquatic environments.

- Evaluate oil-related restoration actions, including development of a ranking scheme to be used in restoration decisionmaking.

- Identify and evaluate tested or promising methods/techniques for non-oil contaminant situations that provide direct insights to oil discharge-related restoration activities for feasibility, effectiveness and success, and costs. Evaluate the applicability of these methods to oil-affected habitats. Evaluate actions as in the above task. The non-oil activities review was, however, limited to approaches that provide direct insights to oil discharge-related restoration actions.

This review is extensive but certainly not exhaustive. A vast literature on restoration and mitigation exists. The authors have attempted to review only information applicable to oil discharges. Additional bibliographies exist. The Restoration Center (NMFS, NOAA, Silver Spring, MD) has developed a directory of restoration experts (Restoration Center, 1996) and maintains a computer database of references (Tim Osborn, contact). There is also a Mitigation Evaluation Data Base maintained by the U.S. Fish and Wildlife Service (USF&WS, Fort Collins, CO, Hamilton and Roelle, 1987, Roelle, 1988). There are several annotated bibliographies focusing on wetland restoration, for example by Schneller-McDonald et al. (1990, also USF&W, Fort Collins, CO). Other sources of information are cited in the following sections, including articles compiled into books and symposia volumes on restoration.

The guidance in this document is meant to summarize existing information and state of the art methods, so that informed decisions can be made in the restoration planning and implementation process. The volume of material presented on restoration reflects more the availability of information than a recommendation to pursue that action.

1.3 Intended Audience

This document was prepared primarily to provide guidance to natural resources trustees using the OPA regulations. However, other interested persons may also find the information contained in this document useful and are encouraged to use this information where appropriate.

1.4 The NRDA Process

The NRDA process shown in Exhibit 1.1 in the OPA regulations includes three phases outlined below: Preassessment; Restoration Planning; and Restoration Implementation.

1.4.1 Preassessment Phase

The purpose of the Preassessment Phase is to determine if trustees have the jurisdiction to pursue restoration under OPA, and, if so, whether it is appropriate to do so. This preliminary phase begins when the trustees are notified of the incident by response agencies or other persons.

Once notified of an incident, trustees must first determine the threshold criteria that provide their authority to initiate the NRDA process, such as applicability of OPA and potential for injury to natural resources under their trusteeship. Based on early available information, trustees make a preliminary determination whether natural resources or services have been injured. Through coordination with response agencies, trustees next determine whether response actions will eliminate the threat of ongoing injury. If injuries are expected to continue, and feasible restoration alternatives exist to address such injuries, trustees may proceed with the NRDA process.

1.4.2 Restoration Planning Phase

The purpose of the Restoration Planning Phase is to evaluate potential injuries to natural resources and services and use that information to determine the need for and scale of restoration actions. The Restoration Planning Phase provides the link between injury and restoration. The Restoration Planning Phase has two basic components: injury assessment and restoration selection.

NATURAL RESOURCE DAMAGE ASSESSMENT
Oil Pollution Act of 1990
Overview of Process

PREASSESSMENT PHASE

- Determine Jurisdiction
- Determine Need to Conduct Restoration Planning

RESTORATION PLANNING PHASE

- Injury Assessment
  - Determine Injury
  - Quantify Injury
- Restoration Selection
  - Develop Reasonable Range of Restoration Alternatives
  - Scale Restoration Alaternatives
  - Select Preferred Restoration Alternative(s)
  - Develop Restoration Plan

RESTORATION IMPLEMENTATION PHASE

- Fund/Implement Restoration Plan

Exhibit 1.1 NRDA process under the OPA regulations.

1.4.2.1 Injury Assessment

The goal of injury assessment is to determine the nature, degree, and extent of any injuries to natural resources and services. This information is necessary to provide a technical basis for evaluating the need for, type of, and scale of restoration actions. Under the OPA regulations, injury is defined as an observable or measurable adverse change in a natural resource or impairment of a natural resource service. Trustees determine whether there is:

- Exposure, a pathway, and an adverse change to a natural resource or service as a result of an actual discharge; or

- An injury to a natural resource or impairment of a natural resource service as a result of response actions or a substantial threat of a discharge.

To proceed with restoration planning, trustees also quantify the degree, and spatial and temporal extent of injuries. Injuries are quantified by comparing the condition of the injured natural resources or services to baseline, as necessary.

1.4.2.2 Restoration Selection

(a) Developing Restoration Alternatives

Once injury assessment is complete or nearly complete, trustees develop a plan for restoring the injured natural resources and services. Under the OPA regulations, trustees must identify a reasonable range of restoration alternatives, evaluate and select the preferred alternative(s), and develop a Draft and Final Restoration Plan. Acceptable restoration actions include any of the actions authorized under OPA (restoration, rehabilitation, replacement, or acquisition of the equivalent) or some combination of those actions

Restoration actions under the OPA regulations are either primary or compensatory. Primary restoration is action taken to return injured natural resources and services to baseline, including natural recovery. Compensatory restoration is action taken to compensate for the interim losses of natural resources and/or services pending recovery. Each restoration alternative considered will contain primary and/or compensatory restoration actions that address one or more specific injuries associated with the incident. The type and scale of compensatory restoration may depend on the nature of the primary restoration action, and the level and rate of recovery of the injured natural resources and/or services given the primary restoration action.

When identifying the compensatory restoration components of the restoration alternatives, trustees must first consider compensatory restoration actions that provide services of the same type and quality, and of comparable value as those lost. If compensatory actions of the same type and quality and comparable value cannot provide a reasonable range of alternatives, trustees then consider other compensatory restoration actions that will provide services of at least comparable type and quality as those lost.

(b) Scaling Restoration Actions

To ensure that a restoration action appropriately addresses the injuries resulting from an incident, trustees must determine what scale of restoration is required to return injured natural resources to baseline levels and compensate for interim losses. The approaches that may be used to determine the appropriate scale of a restoration action are the resource-to-resource (or service-to-service approach) and the valuation approach. Under the resource-to-resource or service-to-service approach to scaling, trustees determine the appropriate quantity of replacement natural resources and/or services to compensate for the amount of injured natural resources or services.

Where trustees must consider actions that provide natural resources and/or services that are of a different type, quality, or value than the injured natural resources and/or services, or where resource-to-resource (or service-to-service) scaling is inappropriate, trustees may use the valuation approach to scaling, in which the value of services to be returned is compared to the value of services lost. Responsible parties (RPs) are liable for the cost of implementing the restoration action that would generate the equivalent value, not for the calculated interim loss in value. An exception to this principle occurs when valuation of the lost services is practicable, but valuation of the replacement natural resources and/or services cannot be performed within a reasonable time frame or at a reasonable cost. In this case, trustees may estimate the dollar value of the lost services and select the scale of the restoration action that has the cost equivalent to the lost value.

(c) Selecting a Preferred Restoration Alternative

The identified restoration alternatives are evaluated based on a number of factors that include:

- Cost to carry out the alternative;

- 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;

- Likelihood of success of each alternative;

- Extent to which each alternative will prevent future injury as a result of the incident, and avoid collateral injury as a result of implementing the alternative;

- Extent to which each alternative benefits more than one natural resource and/or service; and

- Effect of each alternative on public health and safety.

Trustees must select the most cost-effective of two or more equally preferable alternatives.

(d) Developing a Restoration Plan

A Draft Restoration Plan will be made available for review and comment by the public, including, where possible, appropriate members of the scientific community. The Draft Restoration Plan will describe the trustees' preassessment activities, as well as injury assessment activities and results, evaluate restoration alternatives, and identify the preferred restoration alternative(s). After reviewing public comments on the Draft Restoration Plan, trustees develop a Final Restoration Plan. The Final Restoration Plan will become the basis of a claim for damages.

1.4.3 Restoration Implementation Phase

The Final Restoration Plan is presented to the RPs to implement or fund the trustees' costs of implementing the Plan, therefore providing the opportunity for settlement of the damage claim without litigation. Should the RPs decide to decline to settle the claim, OPA authorizes trustees to bring a civil action for damages in federal court or to seek an appropriation from the Oil Spill Liability Trust Fund (FUND) for such damages.

1.5 Basic Terms and Definitions

The term *restoration* is often confused with other similar terms, such as *mitigation*. These various terms are utilized and defined in a variety of ways by various authors. Often the uses of these terms are not very rigorous. For the purposes of the present analysis, it is important to define with some precision what is implied by the term *restoration*.

In the NEPA regulations, the Council on Environmental Quality (1981) provides a broad definition of *mitigation* (Whitaker, 1979):

Mitigation includes:

- Avoiding the impact altogether by not taking a certain action or parts of an action.

- Minimizing impacts by limiting the degree or magnitude of action and its implementation.

- Rectifying the impact by repairing, rehabilitating, or restoring the affected environment.

- Reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action.

- Compensating for the impact by replacing or providing substitute resources or environments (40 CFR Part 1508.20 (a-e)).

The U.S. Fish and Wildlife Service (USFWS) has adopted the above definition of *mitigation* and considers the steps to be in order of desirability in planning (Zagata, 1985). However, in the case of an existing discharge that has caused some impact to a given habitat, *mitigation* would include actions that may be categorized under the third and fifth bullets (i.e., restoration, rehabilitation, replacement, and/or acquisition of the equivalent of lost natural resources or environments). The usage of *mitigation* as including either restoration or replacement actions is a more typical use of the term (Jaworski and Raphael, 1979; Schnick et al., 1982).

In the literature, *restoration* usually refers to actions undertaken to return injured natural resources or services to their baseline condition; that is, at the site (Schnick et al., 1982; Cairns and Buikema, 1984; Cairns, 1988a, 1988b, 1991; Helvey et al., 1991; EVOS-RPWG, 1990b). The term *baseline* is used rather than *predischarge* because, absent the oil discharge, the natural resources may have changed over time creating a baseline different than the predischarge condition. Getter et al. (1984) state that restoration is man's efforts to initiate and/or enhance the recovery process. However, *restoration* is used more generally to include the mitigation actions listed above as provided under the OPA regulations.

Rehabilitation refers to actions that may bring injured natural resources or services to a state different from the predischarge condition, yet beneficial to both the environment and public. This may be necessitated by the fact that it may not be possible to return an ecosystem to the predisturbance condition. For example, species characterizing earlier states of succession may no longer be present, or exotics may be the post-disturbance colonizers (Cairns, 1989). Rehabilitation has also been termed *partial restoration* in that some previously present natural resources and/or functions are restored, while other new, but desirable, ones are introduced. Therefore, rehabilitation may be considered a mix of restoration, replacement, and natural recovery actions (Helvey et al., 1991).

Replacement refers to substituting natural resources or services for those injured. For instance, habitats away from the site of impact may be created or enhanced that provide comparable services in terms of fish and wildlife production (e.g., HEP procedures, U.S. Fish and Wildlife Service, 1980a, 1980b; Schnick et al., 1982; Larson and Neill, 1987; McCollum, 1988). Other lost services provided by natural resources that might be replaced include recreational services, water supply, absorption of nutrients and pollutants (i.e., assimilative capacity), flood and storm damage protection, erosion control, and harvest of natural products (Larson and Neill, 1987; Tiner, 1989).

Acquisition of the equivalent refers to obtaining ownership or other rights to natural resources or services that are comparable to those injured. Typically, it does not involve any direct action on the natural resources themselves, but should be preventative of future impacts, and so be of net benefit.

In the present context, restoration actions performed on-site to facilitate recovery of the affected natural resources will be referred to as *direct restoration*. *Direct habitat restoration* is performed on habitats, while *direct resource restoration* is performed on injured species populations (i.e., fish, shellfish, wildlife). In some cases, direct restoration efforts will actually result in rehabilitation, also assumed to be performed on-site. *Replacement* will refer to actions performed off-site, which serve to mitigate the impact by replacing services lost. The restoration actions described in this document all refer to *primary restoration actions* under the OPA regulations. *Mitigation* or simply *restoration* will be used in a general sense as defined under the OPA regulations.

Except in emergency situations, restoration generally is distinguished from *response* as being performed after the fact and by public trustees, while *response* includes actions performed at the time of the discharge by response agencies (Getter et al., 1984). Response includes containment, cleanup, and protection. Restoration may include physical removal of substrate and vegetation, replanting, and restocking of animal populations.

Definitions under the OPA regulations are contained in Appendix A (§ 990.30 of the OPA rule). Only the more relevant terms are defined below.

1.5.1 Baseline

Baseline means the condition of the natural resources and services that would have existed had the incident not occurred. Baseline data may be estimated using historical data, reference data, control data, or data on incremental changes (e.g., number of dead animals), alone or in combination, as appropriate.

1.5.2 Damages

Damages means damages specified in section 1002(b) of OPA (33 U.S.C. 1002(b)), and includes the costs of assessing these damages, as defined in section 1001(5) of OPA (33 U.S.C. 2701(5)).

1.5.3 Injury

Injury means an observable or measurable adverse change in a natural resource or impairment of a natural resource service. Injury may occur directly or indirectly to a natural resource and/or service. Injury incorporates the terms destruction, loss, and loss of use as provided in OPA.

1.5.4 Natural Resources

Natural resources means land, fish, wildlife, biota, air, water, ground water, drinking water supplies, and other such resources belonging to, managed by, held in trust by, appertaining to, or otherwise controlled by the United States (including the resources of the Exclusive Economic Zone), any state or local government or Indian tribe, or any foreign government, as defined in section 1001(20) of OPA (33 U.S.C. 2701(20)).

Natural resources refer to both habitats (e.g., rocky shores, mud flats, saltmarshes, etc.), and individual biological resources (i.e., animal and plant species, populations, communities, etc.).

1.5.5 Oil

Oil means oil of any kind or in any form, including, but not limited to, petroleum, fuel oil, sludge, oil refuse, and oil mixed with wastes other than dredged spoil. However, the term does not include petroleum, including crude oil or any fraction thereof, that is specifically listed or designated as a hazardous substance under 42 U.S.C. 9601(14)(A) through (F), as defined in section 1001(23) of OPA (33 U.S.C. 2701(23)).

1.5.6 Recovery

Recovery means the return of injured natural resources and services to baseline.

1.5.7 Response

Response (or remove or removal) means containment and removal of oil or a hazardous substance from water and shorelines or the taking of other actions as may be necessary to minimize or mitigate damage to the public health or welfare, including, but not limited to, fish, shellfish, wildlife, and public and private property, shorelines, and beaches, as defined in section 1001(30) of OPA (33 U.S.C. 2701(30)).

1.5.8 Restoration

Restoration means any action (or alternative), or combination of actions (or alternatives), to restore, rehabilitate, replace, or acquire the equivalent of injured natural resources and services. Restoration includes: (a) Primary restoration, which is any action, including natural recovery, that returns injured natural resources and services to baseline; and (b) Compensatory restoration, which is any action taken to compensate for interim losses of natural resources and services that occur from the date of the incident until recovery.

The OPA regulations also include the concepts of primary and compensatory restoration. Primary restoration is any action that returns injured resources and services to baseline conditions, including natural recovery. Natural recovery refers to the taking of no human intervention to directly restore the injured natural resources and services. Depending on the injury of concern, primary restoration actions may include actions to actively accelerate recovery or simply to remove conditions that would make recovery unlikely. For each injury (or loss), trustees must consider compensatory restoration actions to compensate for the interim loss of natural resources and services pending recovery.

1.6 Natural Resources Evaluated

The habitats evaluated in this document are estuarine/marine (saltwater) and freshwater habitats. Exhibit 1.2 lists the habitat categories considered for the evaluation of restoration alternatives and actions within this document.

Exhibit 1.2 is a simplification of the detailed classification system in Cowardin et al. (1979). Cowardin et al. define five major systems: marine, estuarine, riverine, lacustrine and palustrine. The marine and estuarine systems include all waters >0.5‰ salinity (i.e., brackish and saltwater habitats). Riverine, lacustrine, and palustrine are freshwater (<0.5‰) habitats. Riverine habitats are contained within a channel characterized by a flow, either tidally- or gradient-driven. Cowardin et al. (1979) categorize gradient-driven riverine as upper perennial (e.g., brook), lower perennial (e.g., river on a plain), or intermittent. Lacustrine habitats are those situated in a topographic depression or dammed river channel, having less than 30% areal coverage of vegetation, and greater than 8 ha in area (i.e., lakes and ponds). Palustrine habitats are non-tidal freshwater (<0.5‰) wetlands.

Wetland types include emergent, shrub-scrub, forested, aquatic bed, and bog and fens. Emergent wetlands (i.e., marshes) are characterized by erect, rooted, herbaceous perennials. In the marine and estuarine systems, emergent wetlands are the (intertidal) saltmarshes, typically dominated by *Spartina* spp. or *Salicornia* spp. In freshwater systems (riverine, lacustrine and palustrine), marshes contain a diverse assemblage of species (e.g., cattails, rushes, bulrushes, sedges). Shrub/scrub wetlands are freshwater and dominated by woody vegetation less than 6 m tall. Forested wetlands (i.e., swamps) are dominated by woody vegetation greater than 6 m tall. In the marine and estuarine systems, these are mangrove swamps. In freshwater systems, these are hardwood or softwood (coniferous) swamps. Aquatic beds are freshwater wetlands or deepwater habitats dominated by submerged or floating vegetation (e.g., naiads, water lilies). These would be referred to as *weedy shallows* in common parlance. Bogs and fens are dominated by mosses and lichens, and are typically arctic, subarctic, and alpine habitats.

1.7 Possible Restoration Alternatives and Actions

Exhibit 1.3 outlines possible restoration actions under various alternatives, which may be included in a restoration program. Specific actions by habitats and natural resources are provided in Exhibit 1.4. These actions are analyzed in this document. While some of these actions are also used in response, it may be necessary to employ them in the restoration context as well. Therefore, the context under which these actions are reviewed is for restoration, not response. The term *cleanup* is often used as a response action. However, some *cleanup* activities might be correctly considered part of restoration. For this reason, the term *cleanup* will be used in reviewing documentation of response actions, with the understanding that in some situations cleanup techniques are the first step in the restoration process.

The listed actions in Exhibit 1.4 include those with some level of technical feasibility and chance of effectiveness. Those with no viability are not listed, evaluations of feasibility and cost are not provided. However, the reasons for their lack of effectiveness are included in Chapter 3. Inclusion in Exhibits 1.3 or 1.4 does not indicate that the action is recommended in all or any situations. The list is simply to provide organization of the evaluation and discussion.

Exhibit 1.2 Habitat categories considered in restoration guidance.

I. Estuarine and Marine (Saltwater)
- A. Intertidal
  1. Rocky shore
  2. Cobble-gravel beach
  3. Sand beach
  4. Mud flat
  5. Saltmarsh
  6. Mangrove swamp
  7. Macroalgal bed.
  8. Mollusk reef
  9. Coral reef
  10. Seagrass Bed
- B. Subtidal
  1. Rock Bottom
  2. Cobble-gravel bottom
  3. Sand bottom
  4. Silt-mud bottom
  5. Macroalgal (kelp) bed
  6. Mollusk reef
  7. Coral reef
  8. Seagrass bed

II. Riverine
- A. Vegetated (Wetlands)
  1. Emergent wetland (marsh)
  2. Shrub/scrub wetland
  3. Forested wetland (swamp)
  4. Aquatic bed
- B. Non-vegetated
  1. Rock bottom
  2. Cobble-gravel bottom
  3. Sand bottom
  4. Silt-mud bottom
- C. Shoreline
  1. Rock shore
  2. Cobble-gravel shore
  3. Sand shore
  4. Mud shore

III. Lacustrine
- A. Submerged
  1. Rock bottom
  2. Cobble-gravel bottom
  3. Sand bottom
  4. Silt-mud bottom
- B. Shoreline
  1. Rocky shore
  2. Cobble-gravel shore
  3. Sand shore
  4. Mud shore

IV. Palustrine (Wetlands)
- A. Aquatic bed (submerged vegetation)
- B. Emergent wetland (marsh)
- C. Shrub/scrub wetland
- D. Forested wetland (swamp)
- E. Bogs and fens

Exhibit 1.3 Restoration actions for each alternative.

1. Natural Recovery - Monitoring

2. Direct Restoration
   a. Direct Habitat Restoration
      - Contaminant Removal
      - Reconstruction
      - Replanting
      - Accelerated Degradation
      - Monitoring
      - Maintenance
   b. Direct Resource Restoration
      - Restocking
      - Harvest Alteration
      - Enhancement
      - Monitoring
      - Maintenance

3. Rehabilitation
   a. Habitats
      - Contaminant Removal
      - Reconstruction
      - Replanting
      - Accelerated Degradation
      - Monitoring
      - Maintenance
   b. Resources
      - Stocking
      - Harvest Alteration
      - Enhancement
      - Monitoring
      - Maintenance

4. Replacement
   a. Habitats
      - Enhancement
      - Creation
      - Monitoring
      - Maintenance
   b. Resources
      - Reconstruction
      - Replanting
      - Accelerated Degradation
      - Monitoring
      - Maintenance
   c. Non-biological Services
      - Recreational
      - Commercial
      - Cultural

5. Acquisition of Equivalent Resources
   - Acquire Property Rights
   - Protection or Management

6. Combination of the Above

Exhibit 1.4 Possible restoration actions that are evaluated for each habitat and biological natural resource.

I. <u>HABITATS</u>

SALTMARSH
- Natural recovery monitoring
- Low pressure flushing
- Vegetative cropping
- Sediment removal and replacement
- Replanting
- Supplementary erosion control structures
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments
- Wetland enhancement
- Saltmarsh creation

MANGROVE SWAMP
- Natural recovery monitoring
- Low pressure flushing
- Opening of channels
- Replanting (various methods)
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments
- Enhancement
- Creation

FRESHWATER WETLANDS
- Natural recovery monitoring
- Low pressure flushing
- Vegetative cropping
- Sediment removal and replacement
- Replanting
- Supplementary erosion control structures
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments
- Wetland enhancement

Exhibit 1.4 (continued)

BOGS AND FENS
- Natural recovery monitoring
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding

INTERTIDAL MACROALGAL BED
- Natural recovery monitoring
- Vegetative cropping
- Replanting

SUBTIDAL MACROALGAL (KELP) BED
- Natural recovery monitoring
- Vegetative cropping
- Replanting
- Herbivore control
- Kelp bed enhancement (off-site)
- Kelp bed creation

SEAGRASS BED
- Natural recovery monitoring
- Replanting
- Herbivore control
- Seagrass bed enhancement (off-site)

- Reconstruction
- Coral transplants

MARINE AND ESTUARINE ROCKY SHORE
- Natural recovery monitoring
- Flushing (pressure and temperature variable)
- Flushing with chemical remediation
- Sand blasting
- Steam cleaning
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding

Exhibit 1.4 (continued)

MARINE AND ESTUARINE SAND BEACH
- Natural recovery monitoring
- Flushing (pressure and temperature variable)
- Sediment washing
- Agitation
- Sediment removal and replacement
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)
- Incineration

MARINE AND ESTUARINE INTERTIDAL MUD FLAT
- Natural recovery monitoring
- Sediment removal and replacement
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)

MARINE AND ESTUARINE SUBTIDAL ROCK BOTTOM
- Natural recovery monitoring

MARINE AND ESTUARINE SUBTIDAL COBBLE-GRAVEL BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement

MARINE AND ESTUARINE SUBTIDAL SAND BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement
- Capping

MARINE AND ESTUARINE SUBTIDAL SILT-MUD BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement
- Capping

Exhibit 1.4 (continued)

RIVERINE ROCKY SHORE
- Natural recovery monitoring
- Sand blasting
- Steam cleaning
- Flushing (pressure and temperature variables)
- Flushing with chemical remediation
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding

RIVERINE COBBLE-GRAVEL SHORE
- Natural recovery monitoring
- Flushing (pressure and temperature variables)
- Flushing with chemical remediation
- Sediment washing
- Agitation
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)

RIVERINE SAND SHORE
- Natural recovery monitoring
- Flushing (pressure and temperature variables)
- Sediment washing
- Agitation
- Sediment removal and replacement
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)
- Incineration

RIVERINE SILT-MUD SHORE
- Natural recovery monitoring
- Sediment removal and replacement
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)

Exhibit 1.4 (continued)

RIVERINE ROCK BOTTOM
- Natural recovery monitoring

RIVERINE COBBLE-GRAVEL BOTTOM
- Natural recovery monitoring
- Agitation
- Dredging
- Sediment replacement

RIVERINE SAND BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement
- Agitation

RIVERINE SILT-MUD BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement

LACUSTRINE ROCKY SHORE
- Natural recovery monitoring
- Sand blasting
- Steam cleaning
- Flushing (pressure and temperature variables)
- Flushing with chemical remediation
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding

LACUSTRINE COBBLE-GRAVEL SHORE
- Natural recovery monitoring
- Flushing (pressure and temperature variables)
- Flushing with chemical remediation
- Sediment washing
- Agitation
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)

Exhibit 1.4 (continued)

LACUSTRINE SAND SHORE
- Natural recovery monitoring
- Flushing (pressure and temperature variables)
- Sediment washing
- Agitation
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)
- Incineration

LACUSTRINE SILT-MUD SHORE
- Natural recovery monitoring
- Sediment removal and replacement
- Bioremediation:
  - Fertilizer application
  - Oleophilic agents
  - Microbial seeding
  - Tilling of surface sediments (agitation)

LACUSTRINE ROCK BOTTOM
- Natural recovery monitoring

LACUSTRINE COBBLE-GRAVEL BOTTOM
- Natural recovery monitoring
- Agitation
- Dredging
- Sediment replacement

LACUSTRINE SAND BOTTOM
- Natural recovery monitoring
- Agitation
- Dredging
- Sediment replacement
- Capping

LACUSTRINE SILT-MUD BOTTOM
- Natural recovery monitoring
- Dredging
- Sediment replacement
- Capping

Exhibit 1.4 (continued)

II. BIOLOGICAL NATURAL RESOURCE POPULATIONS:

SHELLFISH
- Natural recovery monitoring
- Harvest alteration
- Restocking (various ages)
- Enhancement
- Artificial reefs

FISH
- Natural recovery monitoring
- Harvest alteration
- Restocking (various ages)
- Enhancement
- Artificial reefs
- Stream restoration
- Fish passageway improvement

REPTILES
- Natural recovery monitoring
- Harvest alteration
- Restocking (various ages)
- Enhancement

BIRDS
- Natural recovery monitoring
- Harvest alteration
- Restocking (various ages)
- Enhancement

MAMMALS
- Natural recovery monitoring
- Harvest alteration
- Restocking (various ages)
- Enhancement

TECHNICAL FEASIBILITY OF RESTORATION ACTIONS — CHAPTER 2

2.1 Overview of the Technical Feasibility Assessment

This chapter discusses the technical feasibility of restoration based on information both from actual oil discharge and non-oil restoration situations. It is restricted to technical and engineering issues. Scientific aspects of effectiveness and success are discussed in Chapter 3.

Exhibit 2.1 presents a simplified conceptual overview of potential restoration alternatives and actions. The analysis of technical feasibility was performed for over 30 habitat types. However, conceptually, these habitat types used in Exhibit 2.1 can be categorized as follows:

- Wetlands;
- Biologically structured habitats (e.g., oyster reefs, coral reefs);
- Shorelines; and
- Open water.

The information in this document concentrates on the primary restoration actions for the various habitat types, as well as for categories of biological natural resources (i.e., species groups). Many of the primary restoration actions are also applicable to replacement. For instance, replanting of saltmarshes can be conducted either on- or off-site where an appropriate site exists. Also, habitat enhancement actions may be considered primary restoration actions for the habitats and for individual biological resources that use the habitat.

It was found that a coherent analysis of feasibility required that information from non-oil situations be used to supplement information from oil discharge situations. For instance, saltmarsh restoration has been attempted in few instances after oil discharges. One of the key restoration actions is replanting of the marsh. However, the information available on the few oil discharge restoration attempts is not complete enough to provide an adequate understanding of the full range of factors related to the feasibility of saltmarsh replanting. Thus, the analysis of restoration in saltmarsh habitats includes a specific discussion of cases where restoration was attempted after an oil discharge, but is supplemented with the considerable body of information on saltmarsh replanting that was developed in conjunction with saltmarsh restoration after non-oil injury situations.

Exhibit 2.1 Simplified overview of restoration actions.

General Habitat Types	Habitat Restoration	Habitat Replacement/ Enhancement	Restocking (Primary Natural Resource Restoration)	Habitat Enhancement	Other
Wetlands Saltmarsh Mangrove swamp Freshwater wetlands	• Contaminant removal • Replanting	• Replanting • New wetland creation	• Possible for certain fish reptile and bird species	• Covered under habitat restoration or replacement	• Off-site out-of-kind actions • On or off-site management practices - Harvest alteration - Protecting endangered habitat - Improving recreational services - Preservation - Mitigation banking, etc.
Structured Habitats Vegetated beds Oyster reefs Coral reefs	• Replanting/ reconstruction	• Replanting/ reconstruction	• Limited application	• Generally not feasible	
Shorelines Intertidal Riverine Lacustrine	• Contaminant removal	• Generally not feasible	• Possible for certain birds and mammals	• Limited applicability	
Open Water Subtidal Riverine Lacustrine	• Contaminant removal	• Generally not feasible	• Possible for certain fish species	• Artifical reefs • Stream habitat structures • Fish passageway improvement	

The discussion of technical feasibility includes a description of restoration actions, and consideration of key factors associated with the effective implementation of the action. Factors considered include:

- The general state of feasibility as demonstrated in actual restoration situations;
- The availability of services, expertise, equipment, and materials to perform the action;
- Operational constraints that may inhibit implementation of the action in various situations; and
- The need for future restoration actions, as well as the capability to perform those efforts.

Please note that in this section, consideration of constraints are restricted to operational and technical implementation, not to how effective or successful the action is in the long run. Effectiveness and success is reviewed in detail in Chapter 3. Chapter 2 lays the groundwork for what actions are available for consideration. Effectiveness and success should be the ultimate criteria for choices made.

The technical feasibility of restoration actions contained in this section also takes into consideration the legal and regulatory constraints of the various restoration actions. These factors have a substantial impact on the viability of restoration actions at the site-specific level. At the generalized level addressed in this document, these factors are similar across many of the habitat types and restoration actions. For this reason, the legal and regulatory constraints are presented and key implications summarized in Section 2.5.

The analysis of feasibility of the restoration actions is arranged by habitat type, which are based on the classification presented in Cowardin et al. (1979) (see Section 1). However, some consolidation and rearrangement of the habitat categories was required in order to facilitate a more efficient presentation of the restoration alternatives and actions.

Exhibit 2.2 presents the primary restoration actions by habitat type. These habitats are described in Section 2.2. Section 2.3 discusses restoration of biological natural resources (individual species populations). Section 2.4 evaluates replacement actions (i.e., off-site or out-of-kind). Section 2.5 presents a discussion of legal and regulatory factors associated with restoration

Exhibit 2.2 Primary habitat restoration actions.

Restoration Actions	Saltmarsh	Mangrove Swamp	Freshwate r Emergent Wetland	Freshwate r Scrub-Shrub Wetland	Freshwate r Forested Wetland	Freshwate r Bogs and Fens	Intertidal Macroalgal Bed	Kelp Bed	Eelgrass (Temperate and Subarctic)	Subtropical and Tropical Seagrass Beds	Freshwater Aquatic Beds	Mollusk (Oyster) Reefs
Natural Recovery	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑
Vegetation Cropping	■		■				■					
Replanting	■1	■1	■1	■1	■1		■	■1	■1	■1	■1	
Supplementary Erosion Control Structures	■											
Opening of Channels		■										
Sediment Removal and Replacement												
Off-Site Marsh Creation[2]	■1		■1	■1	■1							
Bioremediation	■					■						
Oyster Reef Reconstruction												■1
Oyster Reseeding												■1
Coral Transplants												
Sand Blasting												
Steam Cleaning		☑										
Flushing (Washing)	■	☑	☑	☑	■							
Sediment Washing												
Sediment Agitation												
Incineration												
Dredging												

Exhibit 2.2 (continued)

Restoration Actions	Coral Reef	Intertidal Rocky Shore	Intertidal Cobble-Gravel Beach	Intertidal Sand Beach	Intertidal Mud Flat	Subtidal Rocky Bottom	Subtidal Cobble-Gravel Bottom	Subtidal Sand Bottom	Subtidal Silt-Mud Bottom	Riverine Rock Shore	Riverine Sand Shore	Riverine Silt-Mud Shore	Riverine Cobble-Gravel Shore	Riverine Rock Bottom
Natural Recovery	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑	☑
Vegetation Cropping														
Replanting														
Supplementary Erosion Control Structures														
Opening of Channels														
Sediment Removal and Replacement					☑		☑	☑	☑			☑		
Off-Site Marsh Creation														
Bioremediation		☑	☑	☑	☑					☑	☑	☑	☑	
Oyster Reef Reconstruction														
Oyster Reseeding														
Coral Transplants	■1													
Sand Blasting		☑								☑				
Steam Cleaning		☑								☑				
Flushing (Washing)		☑	☑	☑						☑	☑		☑	
Sediment Washing			☑	☑							☑		☑	
Sediment Agitation			☑	☑							☑		☑	
Incineration				☑							☑			
Dredging							☑	☑	☑					

Exhibit 2.2 (continued)

Restoration Actions	Riverine Unconsolidated Bottom	Lacustrine Rock Shore	Lacustrine Cobble-Gravel Shore	Lacustrine Sand Shore	Lacustrine Silt-Mud Shore	Lacustrine Rock Bottom	Lacustrine Unconsolidated Bottom
Natural Recovery	☑	☑	☑	☑	☑	☑	☑
Vegetation Cropping							
Replanting							
Supplementary Erosion Control Structures							
Opening of Channels							
Sediment Removal and Replacement	☑				☑		☑
Off-Site Marsh Creation							
Bioremediation		☑	☑	☑	☑		
Oyster Reef Reconstruction							
Oyster Reseeding							
Coral Transplants							
Sand Blasting		☑					
Steam Cleaning		☑					
Flushing (Washing)		☑	☑	☑			
Sediment Washing			☑	☑			
Sediment Agitation	☑		☑	☑			
Incineration							
Dredging	☑				☑		☑

Natural recovery is always an alternative, the "action" as defined in this document being monitoring. In this section, reference to natural recovery as an action implies that monitoring is the only action. Monitoring should accompany all actions. Monitoring (accompanying all actions) is always technically feasible and, therefore, is not described in detail here but is discussed more fully in Chapter 3 (in discussions on each natural resource and in Section 3.2.10).

It should be noted that the distinction between "restoration" and "response" is not always clear. In general, the distinguishing features of restoration are the time period in which it occurs and the government authority overseeing the activities (see Section 1). Restoration occurs in a period of time after the initial response. In some cases, restoration actions analyzed have actually been conducted as part of response efforts, although the basic actions may be applicable to the restoration phase. For example, flushing of shorelines, vegetative cropping, or sediment agitation may be applicable to the restoration phase even though they frequently are conducted as part of an extended response phase. Other response actions are not considered appropriate to restoration activities. These include actions such as sorption and other forms of bulk oil removal.

It must be emphasized that in any restoration situation, individual site-specific conditions will greatly influence the selection of a restoration action. Thus, overall guidance, discussed in this section, should not be interpreted as a detailed step-by-step recommendation in every case.

2.2 Technical Feasibility of Primary Restoration by Habitat

This section discusses the technical feasibility of habitat restoration after an oil discharge by habitat type.

2.2.1 Estuarine and Marine Wetlands

The two major categories of estuarine and marine wetlands are saltmarshes and mangrove swamps.

2.2.1.1 Saltmarshes

Saltmarshes are typically dominated by *Spartina* spp., *Salicornia* spp., *Jaumea carnosa* (Pacific Northwest), or by *Juncus roemerianus*. While the majority of the literature focuses on *Spartina*-dominated marshes, some information exists on other types. Distinctions will be made as appropriate in the evaluations to follow.

Restoration actions developed for saltmarshes include:

- Natural Recovery;
- Replanting;
- Supplementary Erosion Control Structures;
- Sediment Removal/Replacement;
- Vegetation Cropping;
- New Saltmarsh Creation;
- Low Pressure Flushing; and
- Bioremediation.

Other actions may exist under certain situations (e.g., thermal desorption).

Replanting, supplementary erosion control structures, sediment removal (replacement) and vegetation cropping are primary restoration actions. Replanting is a key element in all active marsh restoration and will typically be a component with other actions. Erosion control structures can be coupled with replanting if it is necessary to stabilize the marsh sediment. Sediment emoval/replacement would generally be coupled with replanting. Vegetation cropping is an action that is used to remove residual oil from vegetation that may recontaminate the marsh or contaminate other natural resources. Altering the hydrology of an injured marsh might be considered in extreme cases, and would include many of the considerations under saltmarsh creation.

New saltmarsh creation refers to the development of a replacement marsh at a site different from the injured location. It typically involves hydrological changes and possibly excavation at the new site. It is frequently coupled with replanting using actions similar to those discussed under replanting. Replanting can also be used as an off-site replacement action if a suitable site is available.

Low pressure flushing is most often a response or short-term cleanup action, but it may be part of a restoration to remove residual oil. Flushing is included here because experience with discharges has shown that additional removal of oil may be required even though it is "technically" a cleanup action. Bioremediation is suggested as a potential saltmarsh restoration action but it is still being developed (see Chapter 3).

Saltmarshes are characterized by soft sediments. If marsh vegetation is destroyed, erosion of sediments can readily occur making re-establishment of the marsh difficult or impossible. Injury or alteration of the drainage channels in saltmarshes can affect proper functioning of the marsh.

Considerable injury can occur to saltmarshes as a result of improper restoration. Foot and vehicular traffic can displace sediments and work the oil more deeply into the sediments (Getter et al., 1984; Johnson and Pastorok, 1985; Seneca and Broome, 1982; American Petroleum Institute, 1991). Residual contamination may also be a problem and often complex trade-offs must be made between traffic and residual contamination. These issues are discussed in detail in Chapter 3.

2.2.1.1.1 Oil Related Literature

Restoration of a saltmarsh following an oil discharge is reported in the literature for a limited number of cases. Seneca and Broome (1982) report the results of marsh revegetation efforts in the Ile Grande marsh in Brittany after the *Amoco Cadiz* oil discharge. Krebs and Tanner (1981a) report the results of marsh restoration using a combination of sediment removal, sediment replacement, and replanting in response to an oil discharge in the Potomac River. Mearns (1991) reports on bioremediation in an oiled marsh in Galveston Bay. American Petroleum Institute (1991) evaluates potential restoration using a combination of historical data and *a priori* assumptions. Getter et al. (1984) summarizes information on the restoration of saltmarshes after potential oil discharge injury.

2.2.1.1.2 Non-oil Related Literature

Saltmarsh creation is addressed extensively by the U.S. Army Corps of Engineers including such publications as:

- Army Corps of Engineers (1978) and Woodhouse (1979) - provide design information for creating wetlands using dredged material including extensive guidelines for saltmarsh planting;

- Webb and Dodd (1978) - discuss saltmarsh planting and wave-stilling devices to control erosion in saltmarsh areas;

- Webb and Dodd (1976) - describe early saltmarsh planting projects with the objective of stabilizing marsh shorelines;

- Earhart and Garbisch (1986) - provide detailed discussion of a smooth cordgrass (*Spartina alterniflora*) planting project on a dredged material site;

- Allen et al. (1986) - discuss shore stabilization by planting smooth cordgrass (*Spartina alterniflora*) in combination with temporary breakwaters; and

- Allen et al. (1990) - discuss recent experience with planting saltmarsh species and use of temporary breakwaters to stabilize the shoreline in high wave environments.

Examples of other literature from a variety of marsh restoration efforts include:

- Josselyn and Buchholz (1982) - reports on saltmarsh creation projects in California;

- Havens and Lehman (1987) - discuss results of a saltmarsh creation project as mitigation for construction at a Navy base;

- Allen and Hull (1987) - discuss restoration of a California saltmarsh that had been degraded as a result of urban development;

- Purcell and Johnson (1991) - provide an overview of a degraded saltmarsh that was restored as part of a mitigation project;

- Josselyn et al. (1991) - describe restoration of the Bolsa Chica lowlands in southern California;

- Broome et al. (1988) - summarize their extensive experience with restoring saltmarsh vegetation; and

- National Research Council (1992) - summarizes recent findings and issues on wetland restoration.

Other directly relevant sources of information on saltmarsh creation include Garbisch (1978), Kusler et al. (1988), Josselyn et al. (1990), Broome (1990), Fauer and Gritzuk (1979), Jerome (1979), Zedler (1992), Seneca and Broome (1982), and Seneca and Broome (1992). (Note: the scientific information of these and other literature is reviewed in Chapter 3. These listed sources contain information on technical feasibility.)

2.2.1.1.3 Technical Feasibility of Restoration Actions

Exhibit 2.3 presents a summary of the state of technical feasibility for the alternatives that are discussed in the following sections. Each action should be accompanied by a monitoring program.

2.2.1.1.3.1 Natural Recovery

Monitoring is always a technically feasible action. No other action is associated with this alternative. See Chapter 3 for a discussion of recovery.

Exhibit 2.3 Overview of technical feasibility of saltmarsh restoration.

	State of Feasibility	Availability of Services Materials and Equipment	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery Monitoring	Generally Feasible	Generally available	Little constraint	Replanting or erosion control may be necessary	Coordination of monitoring activities
Replanting	Action has generally been well developed	Specialist restoration firms exist in many areas Experienced labor may be limited Lead time required for nursery plants	Degradation of oil in sediment Tide hampers work Degree of fetch Seeding confined to protected sites Nursery availability for target species may be limited Donor sites for natural propagules	Replanting due to transplant mortality Fertilization	Some states may require permits for gathering propagules
Erosion Control Structures	Generally feasible but varies by site-conditions	Generally available	Large structures require equipment access	Repair Removal	Permits from Army Corps of Engineers and many states

Exhibit 2.3 (continued)

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Sediment Removal/ Replacement	Feasible in only limited circumstances	Readily available in most regions	Possibility of further injury Equipment access	Method may increase injury resulting in extensive additional restoration	Permits from Army Corps of Engineers and many states may be scrutinized
Vegetation Cropping	Generally feasible	Readily available	Possibility of further injury	Collateral injury may result in additional restoration	No formal requirements
New Saltmarsh Creation	Generally feasible, but may require using off-site location	Variable, since projects may range from simple services to massive construction projects	Acquisition of site Establishment of hydraulic regime Availability of suitable substrate Controlling contaminants Pest species	Most viable projects have extensive programs of evaluation and mid-course corrections	Army Corps of Engineers and state agencies have time consuming permit procedures Negotiation for site acquisition
Low Pressure Flushing	Feasible in limited circumstances	Available from oil spill response contractors	Access to marsh interior Possibility of further injury	Additional restoration due to injury caused by the action	No formal requirements
Bioremediation	Action is currently being developed	Services are available from specialists	Few people have strong bioremediation expertise in estuarine and marine systems Possible eutrophication effects	None expected	Permits required

2.2.1.1.3.2 Replanting

Many experts conclude that saltmarsh planting and associated restoration has reached the stage of development where it can be considered a fully feasible method. These experts include Woodhouse (1979), U.S. Army Corps of Engineers (1978), Garbisch (1978), Earhart and Garbisch (1986), Broome et al. (1988), Getter et al. (1984), Seneca and Broome (1992), Josselyn and Buchholz (1982), National Research Council (1992), and others. Zedler (1992) cautions that feasibility is limited to the actual establishment of vegetation that has similar characteristics to control marshes and that full functional equivalence to natural saltmarshes has not been achieved. (See Section 3.2.1 for discussion of effectiveness and success.)

Work conducted by the U.S. Army Corps of Engineers on the large-scale planting of saltmarsh species, which began in the early 1970s, has lead to highly developed replanting actions. In the last decade the proliferation of wetland restoration projects as mitigation for construction has further developed the state of the art. However, there have been few studies that have evaluated the success of these actions, except on the vegetation. (See Chapter 3.2.1.)

Replanting is a prime component of almost all active saltmarsh restoration efforts. The principal methods include:

- Seeding using seed harvested and threshed from a local site;
- Seeding using seed purchased from a commercial supplier;
- Transplanting with sprigs or plugs dug from a nearby saltmarsh site; and
- Planting greenhouse-grown seedlings.

Propagule is a general term for any of various structures used to propogate a plant including seed, seedlings, sprigs, and plugs. Seedlings are small nursery grown plants grown from seed for transplanting. Sprigs are plant stalks with attached roots and rhizome fragments, but with little substrate material. Plugs are plant stalks with a core of intact substrate material, roots and rhizomes.

The planting task can be divided into acquisition of the propagule, and the actual insertion of the propagule into the substrate. Acquiring the propagule from a commercial supplier eliminates the need for including the digging of sprigs or plugs or threshing and harvesting the seed in the scope of the restoration project. However, locally-acquired propagules can be better adapted to the restoration site and may have a higher confidence rate of plant establishment.

Planting can be accomplished using hand methods, semi-mechanized methods using a powered auger, and mechanized methods employing a small agricultural tractor. A limiting factor in the use of the tractor method is the bearing ability of the saltmarsh sediments, accessibility into the marsh, and the size of the project. Mechanized methods may also kill marsh biota left alive through trampling and disruption of root systems (see Section 3.2.1).

Fertilizers are frequently valuable in helping with plant establishment on sandy soils. In other types of soil, they are useful on occasion (Woodhouse, 1979). A slow-release fertilizer can be inserted along with the plug or sprig which will enhance early establishment. Conventional broadcast fertilizers can be applied later during the first year of growth. However, fertilization may interfere with development of the infaunal community and add unnecessary contaminants into the system (see Section 3.2.1).

Smooth cordgrass (*Spartina alterniflora*) is the dominant vegetation in the regularly flooded intertidal saltmarshes on the east and Gulf of Mexico coasts of the United States. Plants that dominate at the higher marsh elevations are saltmeadow cordgrass (*Spartina patens*), saltgrass (*Distichlis spicata*), and black needlerush (*Juncus roemerianus*). Pacific cordgrass (*Spartina foliosa*) is a dominant saltmarsh species in California. Tidal marshes in the Pacific Northwest are typified by such species as, *Salicornia virginica* and *Jaumea carnosa.* Attention must be paid to ensuring that only species indigenous to a specific geographic area are planted. For instance, *Spartina alterniflora* is considered a non-native invasive species on the west coast. *Spartina alterniflora*, *S. anglica*, and *S. patens* are included on the Washington State Noxious Weed List as plant species considered detrimental to natural resources of the state.

Availability of Services, Materials, and Equipment

A number of commercial firms engage in wetland restoration. The growth of such firms was spurred by wetland mitigation projects to offset wetlands lost for construction projects. However, such firms are not widely distributed across the United States. Typically, firms are located in the major areas of the country with broad distribution of *Spartina*-dominated saltmarsh environments. Therefore, restoration activities in isolated areas or where other species dominate may involve considerable travel by specialist restoration firms.

Wetland restoration firms tend to be small specialist operations. A very large restoration project could overwhelm the capabilities of local establishments. Some of the past marsh restoration activities have involved the use of general labor, relatively inexperienced in saltmarsh restoration. Success of the restoration effort is dependent upon experienced supervision by a person knowledgeable in saltmarsh restoration. Some commercial nurseries are beginning to specialize in wetland plant species. However, several months lead time may be required to prepare transplant material. In planning for saltmarsh planting, it is important to coordinate the acquisition of transplant material well in advance of needs.

Constraints

There are a number of operational constraints which may complicate replanting. For example:

- Planting activities must not begin until the oil in the sediment has degraded sufficiently to insure success (see Chapter 3);

- Tides in the saltmarsh environment affect the accessibility to sediments for marsh restoration. Harvesting and planting activities in many locations are confined to a five-hour period per tide, necessitating careful coordination to achieve efficient utilization of personnel and equipment (Woodhouse, 1979);

- Saltmarshes can be established on a wide variety of soils including sand, silt, clay, and peat. Planting is easiest in sand and most difficult in peat; plant growth is usually most effective in silt and clay (Woodhouse, 1979; Broome et al., 1988);

- While seeding is the least expensive method of propagation of saltmarsh species (Section 4.2.1.1.3.2), the use of this action must be confined to protected sites. Seeding is also restricted to the higher elevation areas of the marsh and is limited by seasonality;

- Because of the delicate nature of saltmarsh habitats, foot and vehicular activity in the marsh must be carefully monitored in order to minimize injury. This may be a particular concern if the project is employing relatively inexperienced labor;

- The use of fertilizer may cause concern over eutrophication and encouragement of weed growth; and

- Grazing by herbivores may hinder establishment of planted material.

Future Restoration Actions

Information presented in Broome et al. (1988) suggests that typically about 20 percent of a *Spartina* marsh requires replanting due to transplant mortality. Additional maintenance activities during the first year of marsh establishment include a broadcast application of conventional fertilizer.

2.2.1.1.3.3 Supplementary Erosion Control Structures

Some form of erosion control structure may be necessary in certain instances, such as when the substrate or vegetation is injured to a degree that erosion is a threat. Exposed marshes, where there is a long fetch allowing waves to build, are most vulnerable. Typical erosion control structures suitable for use in saltmarsh restoration projects include:

- Hand-placed slat-type sand fences;
- Small hand-placed sand bags;
- Scrap tire erosion control barriers; and
- Cloth mesh fence.

While large heavy duty sand bags placed with heavy equipment may offer more protection in very exposed situations, their cost is high and the site must have suitable access. Many previously-placed scrap tire erosion control barriers are now being dismantled and their present use may be problematic. Shell cultch can also be used as an erosion control approach (see Section 2.2.4).

<u>Availability of Services, Materials, and Equipment</u>

Woodhouse (1979) reports that slat-type sand fence is available commercially as an erosion control structure. Small sandbags may also be used and installed with hand labor. These materials present no unusual problems in terms of acquisition. The availability of scrap tires varies locally by geographic area.

Larger, more heavily constructed sand bag structures are considerably more expensive and require access routes for heavy equipment (U.S. Army Corps of Engineers, 1978).

<u>Constraints</u>

The large, heavy sand bag erosion control structures are generally limited by accessibility requirements for construction equipment. Such equipment is necessary for filling and placing the structures.

Future Restoration Actions

Periodic repair may be required to maintain effectiveness of the temporary erosion control structures. The devices will have to be removed after the vegetation has established itself sufficiently to stabilize the sediments.

2.2.1.1.3.4 Sediment Removal / Replacement

Krebs and Tanner (1981a) report on the use of sediment removal and replacement as a marsh restoration action. Sediment is removed using excavation equipment such as track-mounted power bucket shovels. When employed, this action would be coupled with replanting as discussed in Section 2.2.1.1.3.2. The primary reason for implementing this is to remove substrate heavily saturated with oil. (See discussion in Chapter 3.)

Availability of Services, Materials, and Equipment

Sediment removal involves the use of readily available construction equipment and services. Firms having the necessary equipment and personnel are geographically far apart. However, because of the significant care that is required to mitigate injury to the marsh, the sediment removal effort should be closely supervised by persons experienced in marsh restoration to prevent unnecessary damage to plants and disruption of the marsh substrate. A "safe" means of disposal is required for the oil-saturated soil. Unfortunately, many locations in the country are located at considerable distance from disposal sites.

Constraints

A major constraint with this action is access to the marsh area by construction equipment. Sediment removal may not be feasible if the sediments consist of fine mud (Getter et al., 1984). This could prevent the conventional excavation equipment from operating in the marsh area. Specialized or "exotic" actions may be available in soft sediments. However, their use in restoration has not been documented.

A related issue is the significant risk of injury to the marsh by equipment and traffic. This action may only be applicable to narrow fringing marshes due to limited access.

Sediment removal without backfilling with clean material lowers the elevation of the substrate and may alter the hydrologic characteristics of the marsh. Thus, sediment removal is only applicable if the substrate slope is relatively steep (i.e., greater than three degrees), otherwise excessive amounts of marsh area could be lost (Krebs and Tanner, 1981a). Also, sediment removal without backfilling may increase the potential for erosion. If backfilling of the stripped sediment is applied, a source of clean fill material must be found. This may be a difficult task at certain restoration sites. Grading of the backfilled area will be required to attain the proper slope and elevation for marsh development.

2.2.1.1.3.5 Vegetation Cropping

Vegetation cropping was performed in a number of cases after oil discharges in saltmarshes. Examples include (Johnson and Pastorok, 1985):

- The *Esso Bayway* discharge in 1979 near Port Neches, Texas;
- The barge *STC-101* discharge in 1976 in lower Chesapeake Bay;
- The *Amoco Cadiz* discharge in 1978 on the coast of France;
- A pipeline discharge in 1974 in Texas; and
- A tank farm discharge in 1976 in the Hackensack River.

While vegetation cropping in these cases was part of the later stages of cleanup activity, the action may be applicable to the restoration phase if heavily oiled marsh vegetation persists. The objective of this action is to remove residual oil that could continue to contaminate the marsh or recontaminate surrounding habitats and biota (such as wildlife).

The general procedure for vegetative cropping consists of manual cutting of the top portions of marsh vegetation using hand tools such as shears, power brush or weed cutters, scythes, or similar devices. After the vegetation is cut, the debris is collected and put into plastic bags for disposal. The work is labor intensive.

This procedure can be injurious to plants. Vegetation cropping typically involves a great deal of pedestrian traffic in the marsh area. This heavy foot traffic has the potential to cause additional injury to the marsh due to trampling vegetation, pushing residual oil deeper into the sediments, disrupting the contour of the marsh substrate, and causing the potential for erosion. In some cases it is feasible to perform the cutting from small boats in the marsh channels. Care must be exercised in the cutting operation to prevent excessive cutting, which may injure the plant root structure (Owens et al., 1992).

The widespread historical usage of this procedure demonstrates the technical feasibility of performing vegetation cropping. This procedure is performed in conjunction with numerous oil discharges in marshes and knowledge of the action is widespread among oil discharge response companies and cooperatives.

Availability of Services, Materials, and Equipment

The method uses general hand labor and small off-the-shelf hand tools. These services and equipment are readily available around the country.

Constraints

This action may have serious problems associated with collateral injury to the marsh. Very soft sediments make access to the marsh difficult on foot and may significantly increase the potential for erosion in the marsh. Erosion can cause extensive injury to the marsh, including loss of suitable substrate and altering of the hydrologic characteristics.

Future Restoration Actions

If this procedure causes additional injury to the marsh, the extent of future restoration actions would increase. Additional injury to the marsh could include further destruction of plants, injury from erosion, and deeper penetration of oil into sediments.

2.2.1.1.3.6 New Saltmarsh Creation

New saltmarsh creation constitutes an off-site replacement type of restoration action. In general, of all wetland types, saltmarsh restoration has been most often attempted. This is attributable to the depth of experience in restoring this wetland type, the ease with which proper elevations can be established (using tide records), and the relatively few plant species that occur in saltmarshes (National Research Council, 1992).

Possible restoration sites could consist of:

- A saltmarsh that was previously degraded due to diking, draining, canals, elevation changes, poor water quality, previous flood control projects, etc.;

- Establishment of a new wetland on a site where disposed dredge spoil has been deposited; and

- Excavation of an upland site.

As early as the mid-1970's, efforts were established to restore injured saltmarshes. One of the largest programs was conducted by the U.S. Army Corps of Engineers. Many state regulations now require mitigation efforts to offset loss of wetlands due to construction projects.

The general actions for creating new saltmarsh involve the following tasks (King, 1991):

- Establish or control the proper hydrology. This may involve:
  - Removing or breaching dikes or levees;
  - Creating tidal channels;
  - Diverting waterflow to or away from site; and
  - Regulating the hydraulic regime, using control structures if necessary;

- Modify substrate, if necessary
  - Excavate to correct elevation and contour
  - Control contaminants
  - Achieve proper soil conditions through fertilization, addition of organic matter, etc.

- Plant vegetation (similar to replanting)

- Monitor progress and make mid-course corrections
  - Monitor marsh productivity;
  - Monitor marsh function;
  - Modify hydraulic regime;
  - Replant; and
  - Control pest species.

Any saltmarsh creation project will involve various combinations of activities that will be highly site-specific. The actual scope of restoration activities will vary significantly depending upon the characteristics of a particular project. Some projects may involve simple breaching of a dike (if the land has not subsided), followed by natural propagation of plants or basic replanting of saltmarsh species. Others may involve extensive re-contouring of site topography using construction equipment. Establishment of the proper hydrological regime may require a complex set of control structures or pumps.

For marshes created on dredge spoil, the concept is similar to marshes created on degraded areas. However, establishment of the substrate at the proper elevation is done by depositing dredge spoil material from U.S. Army Corps of Engineers waterway dredging projects. This frequently requires dewatering and mechanical grading of the spoil material. These types of projects were originally created to find a method for disposal of the dredged material.

Availability of Services, Materials, and Equipment

The requirements for services, materials, and equipment will vary greatly depending on the particular scope of a project. Simple projects will have service requirements similar to those for replanting. For projects with modifications to the topography and hydrology, extensive construction services will be required. This may involve both land-based excavation equipment as well as water-based heavy construction equipment.

For projects established on dredge spoil, the substrate is typically established using water-based barge and dredge equipment. Since these projects are undertaken in conjunction with normal dredging activities, the basic equipment would be available in conjunction with the dredging activities. Grading may require excavation equipment.

Constraints

A significant constraint to the creation of a new saltmarsh is the location of a suitable site. The availability of such sites will vary greatly by location around the country. Another constraint may relate to the establishment of the proper hydrologic regime on the chosen site. If natural flushing does not function effectively at a particular site, a complex series of channels and control structures may be necessary. Some potential sites may be constrained by previous contamination of the substrate. Many degraded wetlands in urban areas are polluted with long-term loadings of toxicants. During the establishment period of the marsh, pest species may invade the site. (This is more problematic on the U.S. west coast than the east or Gulf of Mexico coasts.) Invasive plants may require time-consuming weed removal. Animal pests may require fencing of the area. Insect pests may be problematic and difficult to control.

Future Restoration Actions

This will vary greatly depending on the characteristics of a particular project. To ensure a reasonable chance of success, an extensive program of monitoring and mid-course corrections is required.

2.2.1.1.3.7 Low Pressure Flushing

Low pressure flushing is a technically feasible action in limited circumstances for removing residual oil in marshes. It may not be possible to remove heavily weathered oil without damaging the plant structure and substrate. Typically, engine driven pumps are used to pump water through hoses to flush the oil from contaminated vegetation into marsh channels for subsequent containment and recovery with booms and sorbents. While flushing can be performed from land, it is generally preferred that it be performed from small boats to prevent trampling of vegetation (Johnson and Pastorok, 1985).

Availability of Services, Materials, and Equipment

Equipment and personnel to perform this action are typically available from oil discharge response contractors.

Constraints

Access to the interior of a marsh can be a significant constraint to the use of this action.

Future Restoration Actions

This action may cause further damage to marsh plants and erosion of substrate, thus increasing the need for future restoration actions.

2.2.1.1.3.8 Bioremediation

Bioremediation is a potential technically feasible action for restoration in a marsh. Mearns (1991) reported on the use of bioremediation in an oiled marsh in Galveston Bay. See Section 2.2.6.1.3.5. for a general discussion of bioremediation.

2.2.1.2 Mangrove Swamps

Low-wave energy ecosystems such as mangrove swamps are sites where oil commonly accumulates after a discharge. Mangrove habitats are comprised of complex intertwining root formations that can make the habitats inaccessible, thus hindering the effectiveness of oil removal activities. Restoration actions identified in the literature for affected mangrove habitats include:

- Natural Recovery;
- Replanting;
- Construction of Channels;
- Low Pressure Flushing; and
- Bioremediation.

Replanting can be used as an off-site replacement action, if a suitable site is available. Bioremediation is an action still under development (Scherrer and Mille, 1989).

2.2.1.2.1 Oil Related Literature

Documented restoration projects performed in oil-injured mangrove habitats are identified in Lewis (1981), Lewis (1979), and Mangrove Systems, Inc. (1980). Other literature identifies oil-related impact to mangroves and the necessary activities for restoration (Gilfillan et al., 1981; Getter et al., 1984; Evans, 1985; Teas et al., 1989a,b; and Ballou and Lewis, 1989; Cintron-Molero, 1992).

2.2.1.2.2 Non-oil Related Literature

Mangrove habitats have long undergone stresses from both natural occurrences and man-induced impacts. Injury from natural occurrences includes the impact of hurricanes, natural erosion processes, and tree loss from lightning strikes. Man-induced impacts include stresses related to coastal development and operations such as dredge-and-fill practices.

Literature documenting non-oil related restoration projects involving mangroves include Teas (1977), Goforth and Thomas (1979), and Sosnow (1986). Teas (1977) discusses replanting actions. Goforth and Thomas (1979) detail mangrove restoration for the stabilization of eroding shorelines and replanting activities with the use of small trees. In addition, Sosnow (1986) describes mangrove restoration using seedling plantings in a restoration project performed to mitigate the impacts caused by port dredging activities. Cintron-Molero (1992) recommends natural recovery except in those areas that do not have a ready source of propagules. While these were the primary sources used, there is also a broad body of literature involving mangrove protection and planting, since this was a major issue in South Florida for two decades.

2.2.1.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.4 presents a summary of the state of technical feasibility for each restoration action. Each action includes a monitoring program.

2.2.1.2.3.1 Natural Recovery

Monitoring of natural recovery is always technically feasible. See Chapter 3 for a discussion of monitoring and recovery mangrove swamps.

2.2.1.2.3.2 Replanting

Technically feasible methods of mangrove replanting include the use of, propagules, seedlings or young mangrove trees.

Exhibit 2.4 Overview of technical feasibility of mangrove restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Efforts**	**Legal and Administrative Factors**
Natural Recovery	Generally feasible	Generally available	Little constraint	Replanting may be necessary	Coordination of monitoring activities
Replanting	Demonstrated as feasible under proper conditions	Seasonal availability of propagules Donor sites for trees are limited Specialists exist in many areas	Site elevation Tidal influence Substrate Herbivory Plant quality Residual oil contamination in sediment	Replanting due to transplant mortality	Permits
Construction of Channels	Suggested in literature, but viability not demonstrated	Equipment generally available	Site access for equipment Activities may cause further injury to habitat	Method may cause additional injury requiring further restoration	Dredging permits

Exhibit 2.4 (continued)

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Low Pressure Flushing	Feasible in limited circumstances	Available from oil spill cleanup contractors	Access to marsh interior Excessive soil contamination	Additional restoration due to damage	No formal requirements
Bioremediation	Action is currently being developed	Services are available from specialists	Few people have strong bioremediation expertise in estuarine and marine systems Work crew access needed Possible eutrophication effects	None expected	Permits required

2.2.1.2.3.2.1 Propagule and Seedling Plantings

In this discussion, propagules refer to the seeds or sprouted seeds that are collected directly from mature mangrove trees, or gathered shortly after dropping from trees before exhibiting any additional growth or root formation. Propagules used for mangrove restoration are generally planted or directly inserted into the substrate at a depth of a few inches. Mangrove seedlings are propagules that have germinated and show additional signs of growth such as root development. Seedlings are commonly grown in nursery conditions for a short growing period (3 to 18 months) before they are used for planting material at restoration sites.

The use of both propagules and seedlings is technically feasible as demonstrated in past mangrove restoration projects (Cairns and Buikema, 1984; Getter et al., 1984; Lewis, 1979; Lewis, 1990; Teas, 1977; Teas, 1981; and Teas et al., 1989a,b). For example, just a few years after mangrove propagules were planted in an injured habitat the height and size of canopy developed by the propagules was comparable to that of transplanted, 1-meter high (3 year old) trees (Lewis, 1991). The survival rate of transplanted propagules or seedlings can range from 0% to 100% depending on various characteristics including the action of planting, the type of plant material used, and the planting site.

Propagules are typically the more practical planting method for red mangroves for several reasons. First, propagules are more cost-effective than nursery-raised seedlings. Second, propagules adapt more readily to a habitat because they are not influenced by nursery conditions in which seedlings are raised, offering easier acclimatization to a restoration site. Third, propagules are not as susceptible to injury from wind and other environmental stresses that may blow over top-heavy potted seedlings (Crewz and Lewis, 1991). When planted at the proper elevation in sheltered areas, red mangrove propagules may survive at least as well as older, nursery-grown seedlings (Goforth and Thomas, 1979).

For black and white mangrove species, direct planting of propagules may be impractical, because the propagule must remain on a damp substrate for several days to germinate and anchor properly. Due to the high probability that these propagules may be removed by tides or other influences, planting of black and white mangroves is generally performed using seedlings raised in a nursery environment.

<u>Availability of Services, Materials, and Equipment</u>

Material for propagule plantings is limited to the availability of fresh seeds from mangrove trees. The timing of propagule "drops" is important for restoration planning due to the seasonal availability of this planting material. Red mangrove propagules tend to be available only during a limited period between the summer and fall and cannot be stored for long periods of time. The availability of mangrove propagules in the quantity needed for a planting project typically limits the window of planting opportunity from mid-August to mid-October when the peak fruiting period ends (Getter et al., 1984; Lewis, 1990).

Mangrove seedlings used for replanting injured mangroves are commonly gathered from natural stocks in nearby mangrove habitats or purchased from nursery suppliers. Commercial suppliers of plant material generally have certain species of mangrove seedlings available year-round. If large quantities of certain types of seedlings are needed, longer lead times may be required for contracting nursery plant production.

Mangrove propagules and seedlings are commonly planted by hand using readily available equipment such as boring and digging tools.

Mangrove restoration requires specialized technical expertise to oversee projects. Technical expertise is generally available throughout the regions mangroves inhabit, but may be limited to academic and government scientists and a small number of specialist restoration companies. Most efforts at mangrove restoration noted in the literature where planting actions were employed represented a collaboration of individual expertise that was readily accessed.

Constraints

The following identifies several important factors for planning a mangrove restoration site using propagules and seedlings for planting material.

- Planting Elevation and Slope. For all mangrove restoration sites, the correct intertidal zone elevation must be determined before planting, generally located between mean sea level and mean high water. Elevations depend on the tidal range and should be determined based on the species type to be planted. A survey of existing mangroves at the closest location to the planting is an easy method of determining the correct elevation to plant;

- Tidal Influence/Wave Action. Mangrove plantings are not as viable when performed at restoration sites with high wave energy and tidal influence. The increased wave action can wash the propagules away or disrupt the rooted seedling A review of past plantings in high wave action areas concluded that all attempted plantings, even those at sites with some sort of wave barrier, were not technically feasible due to environmental conditions (Getter et al., 1984). Therefore, a restoration planting site should be one with little or no wave action against the shore to dislodge plantings. Other constraints to performing mangrove planting include stressful environmental conditions such as extremely hot or cold weather, high winds, and low rainfall periods;

- Substrate. There is wide geographic variability in the types of substrates in which mangroves grow. Plantings should be performed in stable substrate, composed of materials such as marl muds and peat mixes. Soil that consists of rock or clay layers may be unacceptable for planting unless it is substantially modified. Sand, clay, or marl substrates may need organic matter added to promote drainage and to support plant and animal colonization, survival, and growth (Crewz and Lewis, 1991);

- Plant Quality. Propagules and seedlings used for planting should be protected from sun and desiccation during transport to the restoration site. For material which is produced by a nursery, the plants should be raised under nursery conditions similar to the conditions at the planting site. Plants destined for saline sites should be raised under a constant salinity regime, not just acclimatized a few weeks prior to planting. When rapid coverage of a site is needed, one- to two-year-old seedlings should be used rather than propagules (Crewz and Lewis, 1991); and

- Sediment Stability. In less stable restoration sites, properly staked, rooted seedlings may be better to use than propagules. Shifting sediments and water movements can easily uproot propagules, while rooted seedlings have a greater chance of survival. In addition, rooted seedlings can also provide greater plant coverage over the short term than propagules and exhibit earlier prop root development for stabilization.

Suitable habitat and environmental conditions are required for maximum growth, survival, and voluntary recruitment of planting material. Primary causes of loss of transplanted propagules and seedlings include:

- Physical removal due to erosion, accumulation of foreign plant material, or floating debris;

- Attacks from organisms such as crabs that eat the seeds, and, in some areas, removal of plants by animals such as rabbits and monkeys;

- Planting at too high an elevation;

- Mortality due to residual oil or other contamination (Getter et al., 1984); and

- Mortality due to natural causes.

In an oil discharge site, the technical feasibility of restoration may be hindered due to residual oil in the sediment causing plant mortality from chronic oil contamination. Some plant material may experience high rates of mortality and some can survive but may develop at a slower growth rate. Additional injury may be imposed on a habitat by excessive human and mechanical intrusion as a result of restoration activity. The use of heavy equipment and steady foot traffic in affected marsh areas could kill existing plant material and prolong soil contamination.

Future Restoration Actions

Additional replanting may be required due to transplant mortality.

2.2.1.2.3.2.2 Young Trees

Mangrove restoration using small mangrove trees as planting material typically involves transplanting nursery-raised trees (approximately 3-5 years old) or trees taken from a nearby donor mangrove habitat. Planting of small mangrove trees from nursery stock provides a potential means of obtaining more rapid growth and substrate stabilization than could be expected from planting propagules or seedlings (Teas, 1977). Mangrove trees were used for restoration projects in both sheltered and eroding or high wave energy areas (Getter et al., 1984; Goforth and Thomas, 1979).

A review of several studies where young mangrove trees were used as transplant material notes that the technical feasibility of using this method has mixed results (Lewis, 1990). Each planting site where trees are used either from donor sites or nursery raised material is unique and transplant viability is primarily the result of actual site characteristics and the type and species of plant material. Survival of transplants using small trees were documented in one report to range from 16% to 100% based on a review of past planting projects (Getter et al., 1984). Factors contributing to transplant mortality included unstable substrate and stress from high wave energy shorelines. Another study indicated that mangrove transplants of 2-3 year old trees had a survival of 98% in 23 months in exposed or high energy areas (Goforth and Thomas, 1980).

Availability of Services, Materials, and Equipment

Small mangrove trees are available year-round from commercial nursery suppliers, although they are generally obtained at a high cost. The availability of trees from donor mangrove habitats is limited due to a lack of available mangroves and increasing concerns for the mortality of mangroves moved long distances or from one region to another. It is recommended by mangrove specialists that plant materials should originate from areas as close as possible to the restoration site (Crewz and Lewis, 1991; Lewis, 1990). Reasons for restrictions on plant material imported from foreign mangrove populations include concerns about transporting "exotic" organisms or diseases between regions, and concerns about diluting the locally adapted genetic stock of mangrove species. Current polices for mangrove site creation and restoration are beginning to restrict the use of plant material from different vicinities.

Required technical expertise on mangrove restoration is available as discussed in Section 2.2.1.2.3.2.1.

Constraints

As described above for propagule and seedling plantings, restoration site conditions are critical to the success of mangrove transplants. Further, the considerations regarding planting elevation and slope, tidal influence and wave action, and substrate quality are equally significant for mangrove trees. For transplanting small trees into a restoration site, the use of plant material from a different ecological zone can affect the reliability of transplants. The tolerance of plant material to restoration site conditions can vary by type of species and care must be taken to properly acclimatize the plant material to the environmental conditions of the transplant site. Plant material that is provided from donor mangrove habitats should come from stock which is native to the region where the restoration site is located. Transport methods and handling of mangrove trees can also affect the viability of the planting effort. Plants should be kept cool, moist, and out of the direct sunlight during transport. In addition, for donor sites, the following guidelines should be met during the transplant procedure:

- Top and side branches should be pruned to two-thirds their original length;
- Trees should be removed with a root ball diameter about half the original tree height;
- The root ball should be watered and stamped down while replacing soil to provide sealing between the root ball and the sides of the hole;
- Trees should be replanted at approximately the same level in the ground and at approximately the same tidal elevation as in the original habitat; and
- Trees should never be planted in unstable substrate.

For all mangrove species, the optimum time period to install mangrove trees is from April to mid-June (Lewis, 1990). Therefore, restoration sites that require construction must be completed prior to the planting window.

Future Restoration Actions

Replanting may be required due to transplant mortality. However, if mangrove trees experienced a high mortality rate after being transplanted and no natural colonization or signs of recovery have occurred, the restoration site may be unsuitable.

2.2.1.2.3.3 Construction of Channels

The construction of channels to increase the level of flushing through a contaminated mangrove habitat was suggested as a restoration action (Ballou and Lewis, 1989). Creating channels may induce flushing (Ballou and Lewis, 1989) and greater habitat circulation (Evans, 1985). However, the literature does not identify specific restoration projects where this action has been demonstrated as successful (see Section 3.2.1).

Availability of Services, Materials, and Equipment

The resources needed to perform construction of channels in an injured mangrove community include materials, equipment, and personnel to perform the desired degree of excavation. This is typical construction equipment that is readily available. Specialized technical expertise to oversee projects is required and is available as discussed in Section 2.2.1.2.3.2.1.

Constraints

Equipment access for the excavation of channels has the potential to be a difficult task depending on channel location. According to Ballou and Lewis (1989), the optimal location for a channel depends on a number of site-specific considerations such as salinity, water levels, and hydrological conditions. Actual siting involves making a site-specific assessment.

A concern regarding the construction of channels as a restoration action is the potential for collateral injury imposed on the mangrove community as a result of this activity. Implementation of channel construction can alter the natural hydrologic conditions of the mangrove habitat (see Chapter 3).

Future Restoration Actions

Future restoration actions may be required due to damage from construction actions.

2.2.1.2.3.4 Low Pressure Flushing

See Section 2.2.1.1.3.7.

2.2.1.2.3.5 Bioremediation

Bioremediation is a potentially technically feasible restoration action in mangrove swamps. Scherrer and Mille (1989) document biodegradation of crude oil in experimentally oiled mangrove soil. See Section 2.2.6.1.3.5 for a general discussion of bioremediation.

2.2.2 Freshwater Wetlands

Restoration of freshwater wetlands, including riverine and palustrine, is similar in concept to the restoration of saltmarshes. However, freshwater wetlands possess some unique characteristics. Exhibit 2.5 summarizes the state of technical feasibility for freshwater wetlands.

2.2.2.1 Emergent Wetlands

Restoration alternatives developed for freshwater emergent wetlands include the following:

- Natural Recovery;
- Replanting;
- Soil Removal/Replacement;
- Vegetative cropping;
- New Wetland Creation;
- Low Pressure Flushing; and
- Bioremediation.

2.2.2.1.1 Oil Related Literature

While there is not an abundance of literature regarding mitigating impacts of oil discharges on freshwater marshes, the following studies document restoration activities following oil discharges:

- Foley and Tresidder (1977) reported on vegetation cropping in response to the NEPCO 140 oil discharge in the St Lawrence River in 1976; and
- Pimentell (1985) reported on restoration including vegetation cropping, sediment removal, and creation of marsh areas adjacent to Little Panoche Creek in Fresno County, California, after a crude oil discharge in 1983.

Exhibit 2.5 Overview of technical feasibility of freshwater marsh restoration.

Restoration Actions	Emergent Wetland	Scrub/Shrub and Forested Wetland	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery	Natural Recovery	Natural Recovery	Generally Feasible	Generally available	Little constraint	Replanting may be necessary	Coordination of monitoring activities
Replanting	Replanting	Replanting	Action has generally been developed	Specialist restoration firms exist in many areas Experienced labor may be limited Lead time required for nursery plants	Degradation of oil in sediment Donor sites for natural propagules Nursery availability for target species may be limited Appropriate elevation and slope Equipment access for tree planting	Replanting due to transplant mortality Elimination of pest species	Some states may require permits for gathering propagules
Soil Removal/ Replacement	Soil Removal/ Replacement		Feasible in only limited circumstances	Readily available	Possibility of further injury Equipment access	Method may increase injury resulting in extensive additional restoration	Permits

Exhibit 2.5 (continued)

Vegetation Cropping	Vegetation Cropping		Generally feasible	Readily available	Possibility of further injury	Collateral injury may result in additional restoration	No formal requirements
New Wetland Creation	New Wetland Creation	Soil Removal/ Replacement	Action has been developed	Variable, since projects may range from simple services to massive construction projects	Acquisition of site Establishment of hydraulic regime Controlling contaminants Pest species	Most viable projects have extensive programs of evaluation and mid-course correction	Permit procedures Negotiation for site acquisition
Low Pressure Flushing	Low Pressure Flushing	Vegetation Cropping	Feasible in limited circumstances	Available from oil spill cleanup contractors	Access to marsh interior	Additional restoration due to damage	No formal requirements
Bioremediation	Bioremedia-tion	New Wetland Creation	Action is currently being developed	Services and equipment generally available	Few people have strong bioremediation expertise in estuarine and marine systems Work crew access needed Possible eutrophication effects	None expected	Permits required

2.2.2.1.2 Non-oil Related Literature

No literature was identified that discussed restoration following discharges of hazardous materials in emergent freshwater wetlands. The following reports discuss restoration in regard to creation of wetlands or the restoration of wetlands previously drained for agriculture:

- Crabtree et al. (1990) describe cases across the country where freshwater marshes were constructed, replanted, and evaluated;

- Bacchus (1989), Clewell (1981), and Willard and Reed (1988) discuss the use of muck/mulch as a seed bank;

- Lee et al. (1976) address various uses of vegetation in conjunction with disposing dredged materials; and

- Piehl (1986), Rondeau (1986), and McCabe and Phillips (1986) address the reclamation of wetlands previously drained for agriculture but being returned to wetland status under a conservation plan. Should these areas be available for wetlands creation, cost-effective creation of new wetlands may be possible.

2.2.2.1.3 Technical Feasibility of Restoration Actions

The following paragraphs discuss the technical feasibility of emergent freshwater wetland restorations actions. Each action should include a monitoring program.

2.2.2.1.3.1 Natural Recovery

Natural recovery monitoring is technically feasible in all cases. See Chapter 3 for an evaluation of recovery with no action.

2.2.2.1.3.2 Replanting

Replanting was used effectively in numerous cases of restoration of emergent freshwater wetlands (typically in response to development permits). Crabtree et al. (1990) describe cases in a number of states where freshwater marshes were constructed, replanted, and evaluated. The widespread historical use of this action demonstrates the overall technical feasibility of replanting efforts. (However, see Section 3.2.2 for discussion of effectiveness and success).

The primary concern from a technical feasibility perspective is the type of species planted in a particular area and availability of the species selected. Lee et al. (1976) reported that the plant species commercially available include *Scirpus robustus* (bulrush), and *Typha latifolia* (cattail). While, the technical feasibility of replanting saltmarsh species is discussed in many literature sources, less has been published on freshwater species. However, examples of the feasibility for various plants exist. Crabtree et al. (1990) describes the following cases:

Location	**Method of Replanting**	**Species Replanted**
Lake Hunter, Florida	Mulching	Pickerelweed, Maidencane, Arrowhead, and Spikerush
Patuxent River, Maryland	Plants and rhizomes	Arrow Arum, Pickerelweed, Arrowhead
Lake George, Minnesota	Topsoil placement	Cattails, Woolgrass, Rushes and Sedges
Rancoas Creek, New Jersey	Plants	Arrow Arum, Arrowhead
Noti-Veneta, Oregon	"Introduced"	Duckweed
Willapa Bay, Washington	Transplanted	Saltmarsh Bulrush, Spike Grass
South Beltline, Wisconsin	Roots and Tubers	River Bulrush, Arrowhead, Burreed
South Beltline, Wisconsin	Mulching	Spike Rush, Aquatic Sedge, Bluejoint Grass, Burreed, Cattail, Lake Sedge
South Beltline, Wisconsin	Plants	Common Reed, Prairie Cordgrass
South Beltline, Wisconsin	Seeds	Smartweed, Marsh Milkweed, Water Smartweed, Marsh Dock, Woolgrass
Kenosha County, Wisconsin	Roots and Tubers	Burreed, Cattail, Arrowhead, River Bulrush, Sweetflag, Smartweed
Kenosha County, Wisconsin	Seeds	Bluejoint, Swamp Milkweed

Replanting may be performed using seeds, roots and rhizomes, propagules, or transplanted species. All types of replanting are used extensively. The seeds, roots, and rhizomes may either be distributed by hand or machine.

A factor considered in replanting is the density of the plants in the initial planting. Lee et al. (1976) reported that population densities of marsh grasses may reach 12,400 plants per hectare. The authors also reported planting *Phragmites communis* to densities equal to 49,400 plants per hectare in diked confinements in the Detroit area. (Note that *Phragmites communis* is a non-native invasive species along the U.S. Atlantic coast and Pacific Northwest.) The authors note that "the spread of most perennial marsh plants is very rapid when conditions for growth are optimal. Given adequate time to autonomously colonize containment areas, the number of propagules introduced could be kept to a minimum."

The feasibility of replanting to restore marshes or create new wetlands has been demonstrated. Although little is published that addresses replanting in response to impacts from discharges of oil or hazardous materials, the ability to transfer technology used on saltmarsh restoration activities to freshwater tidal marshes ensures that technically feasible actions for replanting will be available (see Section 2.2.1.1.3.2).

The use of muck/mulch as a seed bank is common practice (Bacchus, 1989; Clewell, 1981; Willard and Reed, 1988). The hope is that within the muck are seeds, roots, and rhizomes that will germinate or sprout into various indigenous species, effectively replanting the area.

Bacchus (1989) reports on incorporation of a muck layer as follows. Muck is taken from a donor wetland and placed in the new wetland in a layer at least 15 cm thick. This muck layer acts as a seed bank, containing not only seeds from the first wetland's species, but their root and rhizomes as well. This has the potential to allow revegetation of the same species that were present in the donor wetland. In considering the use of muck, impacts on the donor site need to be evaluated.

In general the muck layer reportedly is used for three reasons, it allows rapid reestablishment of a wide diversity of flora not readily available commercially; it simulates substrate conditions (e.g., pH and organic content) existing in the donor wetland, and it establishes beneficial soil microflora and fauna which improve the "vigor of the planted species." Bacchus (1989) additionally discusses various problems in incorporating the muck layer. These include failure of muck to produce perennial marsh species, inhibition of germination or seedling death by interactions with the muck, loss of seed bank effectiveness from storage of the muck, and contamination of the muck with undesirable species (e.g., cattails, primrose willow). Bacchus (1989) presented results from an unpublished study by Dr. Stephen Nielson who found that planting target species in sand substrate was preferable because the presence of a muck layer, even if uncontaminated, is "more conducive to invasion of non-native and nuisance species than sand or clay species."

Clewell (1981) discusses "mulching" using topsoil from natural swamps in connection with vegetation restoration on reclaimed phosphate mines. He recommended the use of topsoil in strips or piles.

Willard and Reed (1988) report on a study by Robertson in which three sites were prepared as follows: One site was left alone as a control, one was covered with "one foot of organic soil (mulch) borrowed from a marsh," and the third site was hand planted with wetlands plants. The mulched site "quickly approached the species richness and density of the donor marsh." The planted site did better than the control, but "suffered from invasion by weed species." Later mulching attempts by Robertson were apparently less viable, leaving the author to conclude that "technical feasibility apparently depends upon the number of propagules of invasive species in the mulch."

Availability of Services, Materials and Equipment

A number of capable, commercial specialist firms engage in wetland restoration. The growth of such firms has been spurred by wetland mitigation projects to offset wetlands lost for construction projects.

If the replanting method uses nursery-raised seedlings, sufficient lead time is required for the nursery to produce required numbers of seedlings.

Constraints

Planting activities should not begin until the contaminant in the sediment has degraded sufficiently to enable success (see Chapter 3).

In the case where propagules or seed bank material are gathered from the wild, suitable donor sites must be available. It is desirable that donor sites be located near the area being restored, to maximize acclimation and minimize logistics.

Pest plant species can be a problem in the restoration of freshwater wetlands. Foot and vehicular activity in the marsh area must be controlled in order to minimize further injury.

Future Restoration Actions

A certain amount of transplant mortality can be expected in a typical restoration planting project. Future restoration actions may include additional replanting.

Maintenance during the restoration project may also be required. Periodic efforts may be needed to eliminate pest plant species.

2.2.2.1.3.3 Soil Removal/Replacement

Removal of soil is performed primarily to remove residues of oil or other hazardous materials that are incorporated into the soils and cannot be removed in any other manner. This soil removal action was used on one riverine wetland in response to an oil discharge. Pimentell (1985) reported on a soil removal effort following a discharge of crude oil into the Little Panoche Creek in California. The soil was removed and stockpiled pending use or disposal. The soil was not replaced. The long-term plan was to allow natural sedimentation to return the marsh to its original state (see also Section 2.2.1.1.3). In effect, the issue of soil removal is not one of feasibility, but rather of doing excessive injury to the remaining habitat and associated costs. (Effectiveness is discussed in Chapter 3, and costs in Chapter 4.)

2.2.2.1.3.4 Vegetation Cropping

Cropping of vegetation is conducted primarily to remove oil residue that adheres to the reeds and leaves and cannot effectively be removed using other methods. Vegetation removal was conducted on two riverine wetlands in response to oil discharges. Foley and Tresider (1977) attempted to use mechanical cutters on the contaminated vegetation, but resorted mostly to hand cutting. Pimentell (1985) cropped vegetation and removed contaminated soil. The technical feasibility of cropping vegetation in an emergent freshwater wetland does not vary greatly from that of cropping in a saltmarsh (see Section 2.2.1.1.3.5.).

2.2.2.1.3.5 New Wetland Creation

The literature regarding the creation of new emergent freshwater wetlands focuses on the following:

- Creation of new wetlands to compensate for other wetlands destroyed by development (e.g., road building);

- Establishment of wetlands in dredge spoil areas; and

- Reclamation of wetlands previously drained for agriculture but being returned to wetland status under a conservation plan.

No literature was found that discusses new wetland creation in response to a discharge of oil or other hazardous material.

Building new wetlands typically requires some excavation to bring the surface level down to the water table, or diking and/or pumping to bring the water level up to the new wetland. Crabtree (1990) reported successful creation of freshwater emergent wetlands across the U.S. The USACOE has demonstrated the feasibility of building marshes in dredge disposal areas. Dikes are used for disposal impoundments to create the proper hydrologic characteristics (see Section 2.2.1.1.3.6.).

Piehl (1986), Rondeau (1986), and others have demonstrated the feasibility of restoring old wetlands that were drained for agriculture to their original state. In many cases, restoration was a simple matter of plugging the fixture installed to drain the water off the area.

Typically, restoration construction operations are coupled with replanting efforts, although the reclamation of drained wetland areas often leaves revegetation to natural recovery. (See 2.2.2.1.3.5. See saltmarsh restoration Section 2.2.1.1.3.2, replanting, and 2.2.1.1.3.6, new wetland creation.)

2.2.2.1.3.6 Low Pressure Flushing

See Section 2.2.1.1.3.7.

2.2.1.3.7 Bioremediation

See Section 2.2.6.1.3.5 for a general discussion of bioremediation.

2.2.2.2 Scrub-Shrub Wetlands

Restoration alternatives developed for scrub-shrub wetlands are the same as those for forested wetlands (See Section 2.2.2.3.) and include:

- Natural recovery;
- Replanting;
- New Wetland Creation;
- Flushing; and
- Bioremediation.

Most of the literature discussing restoration of freshwater wetlands dominated by woody plants focuses on forested wetlands. The restoration of a scrub-shrub wetland is very similar to restoration of a forested wetland (with the exception that shrubs rather that trees are the vegetation of choice). The technical feasibility of restoration of forested wetlands is considered in Section 2.2.2.3.

2.2.2.3 Forested Wetlands

Forested wetlands vary from wooded swamps to bottom land riparian habitats. Wooded swamps occur primarily in floodplains or shallow lake basins. Their soils are saturated to within a few inches of the surface or covered by several feet of water. The wetland may be flooded occasionally, seasonally, or for much of the year. Vegetation ranges from the water-tolerant wooded swamp varieties to typical bottom land species (e.g., cypress, tamarack, red oaks, gums). These characteristics affect the choice of actions for wetland restoration.

Restoration actions developed for forested wetlands include:

- Natural Recovery (monitoring);
- Replanting;
- Forested Wetland Creation;
- Low-Pressure Flushing; and
- Bioremediation

2.2.2.3.1 Oil Related Literature

No information was identified on restoration efforts in response to an oil discharge.

2.2.2.3.2 Non-oil Related Literature

The following reports discuss technical feasibility of restoration in non-oil discharge situations:

- Posey et al. (1984) provide information regarding upland and wetland creation and restoration at the Ravenwood shellrock mine. The discussion includes use of large tree spade for transplanting of adult trees;

- Brown et al. (1984) provide information regarding wetland reconstruction following phosphate mining, especially regarding the preparation of a peat substrate and vegetating with wetland species;

- Landin (1982) - This U.S. Corp of Engineers (USACOE) report discusses the restoration of mining lands in Louisiana to forested wetlands after regrading using local, water tolerant species;

- Weston and Brice (1991) discuss the restoration of hardwood wetlands after invasion by the exotic species. The exotic species, Brazilian pepper, was removed and the area replanted with indigenous species;

- Willard et al. (1990) provide information regarding restoration of riparian wetlands in the Midwest. The study primarily addresses restoration management (i.e., siting restraints, revegetation specifications, and long-term vegetation management requirements); and

- Jensen and Platts (1990) focus on the restoration of degraded riverine/riparian habitat in the Great Basin and Snake River regions.

2.2.2.3.3 Technical Feasibility of Restoration Actions

The technical feasibility of restoration actions is discussed below. Each action should include a monitoring program.

2.2.2.3.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible in all cases. See Chapter 3 for a discussion of recovery potential.

2.2.2.3.3.2 Replanting

Replanting a forested wetland may be accomplished by seeding, planting seedlings, planting cuttings, and transplanting adult trees.

<u>Planting from Seed</u>

Direct seeding can be used in restoration projects. McElwee (1965) noted that direct seeding is cheaper than transplantation and the effects of "root disturbance" are eliminated. At the time of the author's report there existed many uncertainties to seeding, including site preparation, collection and storage, sowing rates, and protection from rodents.

Seeds must be collected when ripe and may require preparation prior to planting (i.e., soaking, scarifying, temperature treatment, etc., Willard et al., 1990). Seeds may be broadcast from the ground, boats, or aircraft.

In a report discussing seeding of oaks with acorns, Johnson and Krinard (1987) gathered acorns and collected information regarding seed handling, planting methods, survival, and competition. They noted that "sowing in the winter generally produces the best results" with one possible explanation being less loss to rodents. They note that satisfactory results were achieved from summer plantings and monthly plantings. The study notes that the major reasons for seeding failure are "flooding, droughts, residual herbicides, poor quality seeds, and animal damage."

Planting Seedlings

The technical feasibility of planting seedlings (young plants grown for transplanting) in various sizes (typically measured in gallons of the root ball) is recognized both commercially, i.e., nurseries regularly sell and plant such items, and in the literature regarding wetland restoration. Clewell (1981) noted in a study of restoration of reclaimed mine lands that the planting of seedlings is technically feasible for forest reestablishment and considered inexpensive so long as a mechanical tree planter is used. Landin (1982), in discussing the creation of a wetland on a dredge disposal site in Texas, also noted the technical feasibility of transplanting seedlings. Denton (1990), in a study of the growth rates and planting recommendations for cypress trees at forest mitigation sites, reports that this study found no difference in the survivorship of one-, three-, or seven-gallon trees.

Weston and Brice, (1991) discusses planting of species indigenous to central Florida following removal of an exotic species. The species planted were from a local nursery and hand-planted using unskilled labor from a non-profit youth organization. The species planted on the one hectare plot, their root ball size, and their survival rate are shown below.

Examples of Species	Root Ball Size	Number	Survival Rate
Red Maple (swamp area)	10 gallon	25	70%
Pond Cypress (swamp area)	15 gallon	38	98%
Pond Cypress (pond area)	15 gallon	20	98%
Pop Ash (pond area)	15 gallon	10	(na)
Swamp Tupelo (swamp area)	3-5 gallon	17	66%

Planting Cuttings

The use of cuttings (i.e., branches cut and planted without root growth) from various species to revegetate forests during wetland creation is documented. Jensen and Platts (1990) reported in a case study of a wetland created in Idaho, that willow cuttings used in restoring a riparian habitat had the "about the same" survival after one season as rooted stock. Available moisture in the soil was reported more important than the method of propagation. Cuttings of some species were found to survive better than others. Willow cuttings out-performed cuttings from some understory species used at the same site (e.g., choke cherry and dogwood). Carothers et al. (1990) reported using cottonwood and willow cuttings that were rooted at a nursery, bagged in one-gallon root balls, and used for wetland creation and restoration. They also discussed planting cottonwood and willow poles (cuttings) that were four to 20 feet long, cut from dormant living trees. (Non-dormant poles from which all leaves were removed could also be used.) The bases of the cuttings were "scored with an axe and dipped in a fungicide/hormone solution," after which they were buried in saturated soil.

Transplanting Adult Trees

Clewell (1981) discussed transplanting trees from natural swamps to reclaimed mining lands with a tree spade. The author noted that "tree spading of saplings up to 8 cm in diameter can be accomplished, though often with limited success." He noted that the operation is limited to soils firm enough to support the equipment.

Posey, Goforth, and Painter (1984) documented the feasibility of transplanting large, adult trees to a wetlands creation site using a large tree-spade. The study, located in central Florida, used a "Big John 78 Tree Spade" with a capacity to collect a 3,400 kilogram root ball with a two meter diameter. The authors report that trees to a height of nine meters could be cost-effectively transplanted using this method. The following tree species were transplanted with the tree spade.

The trees used in this study were taken from an adjacent area scheduled to be cleared and strip mined. Availability of indigenous species for transplant will vary depending on presence of trees on a donor site.

Carothers, Mills, and Johnson (1990), noted that "mature trees of any size can be boxed and moved." They note that while this action was used to salvage trees in areas to be developed, the action has not been used in restoration or creation projects. They state that "in some cases this action may be useful" but they note that cost is "its main drawback." The procedure requires pruning to reduce transpiration, digging trenches on all sides, building a box, watering for about two weeks, and cutting any tap roots followed by installing the box bottom. Maintenance (e.g., watering if the ground is not saturated) may be performed indefinitely. The authors report that survival rates average over 90 percent, regardless of tree size. The authors listed several species transplanted including mesquite, paloverde, ironwood, ash, willow, and various shrubs.

Species (Zone within Wetland)	Size (Height)	Number Moved	Survival Rate
Slash Pine (buffer zone)	15-30 feet	1,050	88%
Sable Palm (buffer zone)	15-35 feet	350	97%
Bald Cypress (littoral zone)	15-30 feet	80	75%
Pond Cypress (littoral zone)	5-8 feet	60	89%
Red Maple (littoral zone)	10-15 feet	30	86%
Red Maple (buffer zone)	15-25 feet	36	91%

2.2.2.3.3.3 Forested Wetland Creation

Many studies reported in the literature discuss the technical feasibility of creating a new forested wetland. The lands used for the new wetland range from natural stream or riverside areas (Bacchus, 1989; Willard et al., 1990; Jensen and Platts, 1990) to old strip mines (Posey et al., 1984; Brown et al., 1984; Landin, 1982).

Critical aspects of planning the creation of the forested wetland (Willard et al., 1990) include construction, hydrology, substrate, revegetation, fauna reintroduction, buffers, and long-term management. These items are discussed below.

Construction - Excavation (including removal of contaminated soils), contouring, and channel construction may be necessary to prepare a non-wetland area for a forested wetland (Willard et al., 1990). Contouring was used on old mine lands to return the topography to that of the land prior to strip mining (Jensen et al., 1990). As discussed by Willard et al. (1990) timing of the construction should be managed so as to minimize exposure of open ground subject to erosion.

The removal of exotic pest species prior to wetland restoration (Weston and Brice, 1991) or other nuisance species during restoration (Bacchus, 1989) are examples of preparation of the land prior to replanting. One study performed in Florida discussed the removal of an exotic species prior to replanting with indigenous species (Weston and Brice, 1991). Trees were cut with chainsaws and removed by hand. The vegetation was hauled to a waste recovery plant. All cut stumps were treated with a herbicide to stop regrowth. Felling of large trees may be accomplished by chainsaw, but will require full scale timber operations including skidders to haul out timber and logging trucks with lift arms to pick up and remove the logs.

Hydrology - Since wetland communities are "determined by hydrology," managing water levels is important. Willard et al. (1990) indicate a preference for natural site hydrology. However, permanent, low-maintenance water control structures such as levies or channels may be useful.

Substrate - The substrate is important in supporting the desired wetland functions (e.g., water retention) as well as supporting the desired vegetation (e.g., nutrients, compaction). Willard et al. (1990) point out that substrate can be altered by soil removal and/or replacement. Actions involving removing or modifying soils include:

- Off-site peat is brought in and used as a substrate (Brown et al., 1984);
- Off-site muck is brought in for substrate (Bacchus, 1989);
- Clay or silt may be added to a porous substrate in order to slow percolation (Kobriger et al., 1983); and
- Fertilizer should be added only to those substrates that are very infertile (Kobriger et al., 1983).

It should be noted that adverse impacts may occur to existing functioning wetland systems when they are mined for their substrate. This method should only be employed when substrates are collected from sites that are already slated for development or other adverse impacts. The objective of amending soils can also be achieved through the incorporation of organics (such as sterile straw or other commercially available products) into existing substrate. The use of peat should be avoided as the mining of these systems has resulted in their regional scarcity.

Revegetation - The proper vegetation selection is critical to the restoration effort. Typically with the creation of forested wetlands, an annual ground cover is established within which trees are planted. Timing is critical since replanting should be accomplished in the proper season to ensure high survival and first-year growth (Willard et al., 1990). Replanting is discussed under Option B above.

Reintroduction of Fauna - Typically in forested wetlands creation, fauna are allowed to recolonize naturally. Willard et al. (1990) note, however, that this passive reintroduction will only work if there are "adequate corridors to allow movement between existing populations and the project site."

Buffer Areas - Willard et al. (1990) state that "buffers are an essential component of wetland systems." These buffers serve to protect the new wetland from "outside disturbances" and act as corridors for floral and faunal reintroduction. The size of buffers needed depends on the nature of adjacent development or habitats.

Long-Term Management - Restoration must have a long-term management plan to achieve success (Willard et al., 1990). Vegetation management through mechanical control or controlled burnings is the most common form of long-term management. Willard et al. (1990) reports that managers often "wish to dredge wetlands." Dredging can significantly affect wetlands. The authors recommend either evaluating and modifying water control to flush sediments or accepting accumulation as a natural part of wetland dynamics.

2.2.2.3.3.4 Low Pressure Flushing

See Section 2.2.1.1.3.7.

2.2.2.3.3.5 Bioremediation

See Section 2.2.6.1.3.5 for a general discussion of bioremediation.

2.2.2.4 Bogs and Tundra

Bog type ecosystems in the U.S. are typified by the northern peatlands in Wisconsin, Michigan, Minnesota, and the glaciated Northeast (Mitsch and Gosselink, 1986). Similar peat deposits are found in the Pocosin area along the Virginia and Carolina coasts. Bogs are found in the Appalachian mountains of West Virginia. The tundra ecosystem in Alaska is similar to bogs because of low water interchange and similar characteristic vegetation (e.g., mosses).

Most bog ecosystems are the final stages of the "filling-in" of old lake basins formed from glacial activities. The centuries of debris deposited in basins forms the peat substrate that characterizes these systems. Bogs are characterized by a lack of nutrients and waterlogged, anaerobic, low pH conditions (Mitsch and Gosselink, 1986).

In Europe, late-stage marshes are classified as fens. Fens are characterized by more open waters, more nutrients, and "marsh-like vegetation" such as grasses, sedges, or reeds. The fens are transitional stages between marshes and bogs, but are, as noted by Mitsch and Gosselink (1986), classified as marshes under North American terminology.

The lengthy development time of the peat deposits in bogs is an important characteristic to understand in assessing human limitations in restoring affected bog systems. Hammer (1982) notes, in *Creating Freshwater Wetlands*, that efforts to establish bogs should begin by establishing marshes, which are successional stages to bogs.

Restoration actions presently available for bogs include:

- Natural Recovery; and

- Bioremediation.

2.2.2.4.1 Oil Related Literature

No information was identified on bog restoration efforts in response to an oil discharge. Brendel (1985) presented results from various restoration attempts for oil discharges on tundra around a trans-Alaska pipeline check valve.

2.2.2.4.2 Non-oil Related Literature

No information was identified on restoration efforts of bog ecosystems in non-oil situations.

2.2.2.4.3 Technical Feasibility of Restoration Actions

Actions considered include natural recovery and bioremediation, as discussed below.

2.2.2.4.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible in all cases.

2.2.2.4.3.2 Bioremediation

Brendel (1985) reported on bioremediation attempts on tundra. The comparative analysis was conducted over a three-year period and involved various combinations of tilling, fertilizing, seeding, and bacteria placement (i.e., bioremediation). Bioremediation in the future may be considered as an action for restoring affected tundra (and possibly bogs). Presently, bioremediation is not fully developed, and, therefore, is not a feasible action in tundra or bog habitats. See Section 2.2.6.1.3.5 for a general discussion of bioremediation.

2.2.3 Vegetated Beds

Vegetated beds are classified as estuarine and marine macroalgal, seagrass, and freshwater aquatic beds. Macroalgal beds are classified as intertidal and subtidal (i.e., kelp) beds. Seagrass beds include temperate (e.g., *Zostera* spp. referred to as eelgrass, *Ruppia maritima*), subtropical, and tropical seagrass beds.

2.2.3.1 Macroalgal Beds

This section discusses intertidal macroalgal beds and kelp beds.

2.2.3.1.1 Intertidal Macroalgal Beds

Intertidal macroalgal beds occur on rocky and cobble intertidal areas. No documented case of restoration of intertidal macroalgal beds was identified. However, American Petroleum Institute (1991) suggests a possible scenario for transplantation in this habitat involving reestablishment of selected organisms, i.e., algae and selected fauna. This transplantation method includes collection from a suitable unaffected nearby area, transportation to the cleaned discharge site, and establishment at the restoration site. This is currently a rather speculative process, since little actual field implementation is documented in the literature. As with any transplanting activity, the effect on the donor site would need to be considered.

As intertidal macroalgal beds occur on rocky and cobble shorelines, technically feasible actions for rocky shores and cobble-gravel beaches are also feasible here. Considerations for the choice of actions will include evaluation of further injury caused by the action.

2.2.3.1.2 Kelp Beds

Most of the literature on kelp bed restoration focuses on those habitats dominated by the large brown alga *Macrocystis pyrifera* (Schiel and Foster, 1992). This habitat sustains a large number of dependant species. Restoration actions identified in the literature for injured *Macrocystis* kelp beds include:

- Natural Recovery;
- Replace with Transplants; and
- Vegetation Cropping.

Replacement can be used as an off-site replacement action if a suitable site is available. It should be noted that little or no research has been documented on other types of kelp beds (e.g. *Nereocystis*, *Laminaria*) and it is not known how applicable these actions are to these other systems.

2.2.3.1.2.1 Oil Related Literature

The available literature does not document any restoration attempts of subtidal kelp habitats performed due to oil contamination.

2.2.3.1.2.2 Non-oil Related Literature

Schiel and Foster (1992) describe attempts at kelp restoration. Historical restoration attempts identified in this paper include both "trial and error" experiments as well as more refined studies and applications of scientific actions. For example, many kelp habitat improvement projects were directed at restoring or expanding kelp forests in California over the past twenty years. Numerous unpublished reports were produced to document these efforts in regions including Los Angeles, San Diego (Point Loma), and Santa Barbara. Joint studies and restoration projects were conducted by the California Department of Fish and Game (CDFG) and Kelco Company, the largest kelp harvesting company in the state of California. Since 1987, the focus has been on injured kelp habitats in Santa Barbara among other regions in southern California (Schiel and Foster, 1992). Selected publications that review these restoration attempts are referenced in Schiel and Foster (1992). Kelp mitigation projects are also underway in the San Diego region as a result of kelp depletion by the San Onofre Nuclear Generating Station (California Coastal Commission, 1991). Another report documents restoration methods used to restore storm-injured kelp beds (CDFG, 1990).

2.2.3.1.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.6 presents a summary of the state of technical feasibility for the actions considered for kelp bed restoration. Each action should include a monitoring program.

2.2.3.1.2.3.1 Natural Recovery

Monitoring of natural recovery is a technically feasible restoration action. The ability of an injured kelp bed to recover is discussed in Chapter 3.

2.2.3.1.2.3.2 Replace with Transplants

Transplanting was demonstrated as technically feasible for non-oil related injury to kelp beds. Transplanting involves the use of replacement substrate and plant material. Variations of this action include:

- Using mushroom anchor artificial growth centers (AGCs);
- Using mushroom anchor AGCs with transplants; and
- Stapling loose plants.

Exhibit 2.6 Overview of technical feasibility of kelp bed restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Efforts**	**Legal and Administrative Factors**
Natural Recovery	Generally feasible	Generally available	Little Constraint	Replanting may be necessary	Coordination of monitoring activities
Replace with Transplants	Demonstrated as feasible under proper conditions	Specialist restoration expertise is required May be lag-time for anchor construction	Availability of spore population during deployment Unsuitable habitat conditions for planting Herbivory	Replanting due to transplant mortality	Permits
Vegetation Cropping	Demonstrated as feasible under proper conditions	Available	Possibility of additional injury	Possibility of collateral injury	No formal requirements

Artificial Growth Centers

Concrete anchor devices are used as artificial growth centers for kelp development in injured habitats. The anchors are placed on the substrate to attract plant spores. "Mushroom" concrete anchors are designed with a convex bottom and a flat top surface. This design provides a surface on which macroalgal spores can attach themselves in the absence of other suitable bottom substrate (e.g., rock). To stabilize spore attachment and growth, the anchors are fitted with rebar material (e.g., handles) that is set in the concrete. These "handles" help to secure a growing plant to the anchor. The mushroom anchors are deployed from a vessel with the use of a steel pole attached to tubing in the concrete anchors. The anchors are placed on the bottom. In sand bottom environments, the anchors are buried so that only the flat side of the anchor is exposed (CDFG, 1990). In northern, protected waters (i.e., Puget Sound), the placement of less sophisticated substrate (i.e., large rock, boulders) on otherwise featureless bottoms has proved suitable for kelp holdfasts.

Artificial Growth Centers with Transplant Material

The use of artificial growth centers (AGCs) with juvenile kelp plants (transplants) may accelerate recovery of an injured habitat compared to the use of anchor AGCs alone. This has been demonstrated as a technically feasible approach to kelp restoration (CDFG, 1990; Schiel and Foster, 1992). The plant material is secured with the use of a special type of wire that is attached to the anchor surface. The rebar handles offer support for the transplants. Transplant material provides an additional source of natural spores for recolonization as well as an immediate habitat for other organisms. Transplants are obtained by laboratory growth of plant spores to a desired development stage, followed by "outplanting" to the field. It is necessary for these plants to reach over 1 meter in height before they can be placed in the environment.

Staple Loose Plants to Habitat Bottom

Another demonstrated action for restoration of injured kelp beds involves securing loose plant material to the habitat bottom using large metal staples. This action was used in sandy bottom environments (e.g., California). Based on available literature, the best method for securing loose plant material is the use of two-foot long rebar staples with hose "barbs" attached to the ends (CDFG, 1990). The staples are driven through the loose plant into the substrate. The barbs provide a secure hold on the plants. This action was demonstrated as technically feasible when used in environments with soft bottom material. This approach, however, may not be as feasible in hard bottom kelp bed habitats.

Availability of Services, Materials, and Equipment

For the actions described above, the materials and equipment required can, in general, be easily obtained assuming that restoration takes place in an area close to boating and transportation suppliers. An exception to this may be the availability of concrete anchors, which have to be constructed by a manufacturer. In addition, the availability of transplant material for attachment to concrete anchors will depend on the capability of local plant nurseries to supply the required material.

The majority of kelp restoration projects documented in the literature are located in California. As a result, specialist personnel experienced in the transplant actions described above are concentrated in this region. Schiel and Foster (1992) outline a comprehensive bibliography of studies on the kelp community, identifying a number of technically qualified persons who could oversee restoration. The placement of anchors and transplant material also requires vessel operators, divers, and other technical personnel. These labor requirements can be generally fulfilled in coastal communities.

Constraints

In general, restoration should occur in areas where kelp grew in the past. In planning restoration, the desired growth density should be chosen so that it is in within normal range (i.e., that which is observed naturally in the region). When artificial growth centers are used, the most effective time period for deployment is from September to December, the peak colonization period for *Macrocystis* spores. Anchors with transplants should be used where macroalgal spores are not available for recruitment, such as in the late winter and spring when species other than kelp might colonize the growth centers. When anchors are used in sand bottom habitats, it is important that the anchor not bury completely in the sand so that an exposed surface is available for new algal spores to develop. This may be prevented by the use of heavier anchors (e.g., 45-65 pounds each), which are better able to stand up to wave surges and other forces that may cause burial. Further, heavier anchors are able to secure the largest plants expected to develop within a year from deployment.

Suitable habitat and environmental conditions are required for maximum growth, survival, and voluntary recruitment of planting material. The technical feasibility of planting activities is hampered by high sedimentation, which prevents light and nutrients from reaching the plants, high water temperature, high levels of turbidity, which can scour and leave abrasions on the plants and prevent macroalgal spores from colonizing on substrate, and poor quality substrate, which can affect the character of algal stands.

Future Restoration Actions

Future restoration actions may be needed (e.g., additional transplants) if recovery is slow.

2.2.3.1.2.3.3 Vegetation Cropping

Vegetation cropping of oiled *Macrocystis* stands has been documented as technically feasible for removing residual oil from kelp beds in the context of cleanup operations (Johnson and Pastorok, 1985; API, 1991). However, there are no documented cases where kelp vegetation cropping has been performed in either oil discharge- or non-oil discharge related restoration projects.

Vegetation cropping has the potential to cause further injury to the habitat. However, this injury can be mitigated by taking certain actions in conjunction with the cropping operation. These include, leaving untouched kelp strips among the clear cut areas, harvesting only the minimum length of kelp necessary, and selective thinning of kelp plants (Johnson and Pastorok, 1985; API, 1991).

Vegetation cropping is only appropriate where the macroalgal species involved, such as Macrocystis spp., does not grow from the tip of fronds. This needs to be evaluated before considering this action for other species.

2.2.3.2 Seagrass Beds

Seagrass beds in the U.S. may be classed as either temperate and subartic or subtropical and tropical. Eelgrass (*Zostera marina*) is in most cases the dominant species of temperate and subartic beds, extending from near the Arctic circle on both coasts of North America south to North Carolina on the east coast and to the Gulf of California on the west coast. Dominance by *Ruppia maritima* is also common (worldwide). In the subtropical and tropical climatic regions (i.e., Florida, the Gulf of Mexico, and the Caribbean), several types of seagrass are found. The dominant species in these regions include turtlegrass (*Thalassia testudinum*), Cuban shoalgrass (*Halodule wrightii*), and manatee grass (*Syringodium filiforme*).

Identified actions for seagrass restoration include:

- Natural Recovery (monitoring); and
- Replanting.

Replanting can be an on-site restoration action or an off-site replacement action, if a suitable site is available.

2.2.3.2.1 Oil Related Literature

Based on a search of published literature and communications with technical experts, there are no documented cases where seagrass habitats injured by oil contamination have been restored (Zieman et al., 1984; Fonseca, 1991; Thayer, 1991).

2.2.3.2.2 Non-oil Related Literature

Seagrass restoration is extensively documented in the literature for non-oil related habitat impacts. These publications include:

- Thorhaug and Austin (1976) discuss results of historical eelgrass projects including methods used and habitat conditions;
- Fonseca et al. (1979) summarize results of a restoration effort performed in an eelgrass habitat injured from scallop dredging;
- Phillips (1980) provides restoration planting guidelines for various types of seagrass restoration;
- Thorhaug (1980) describes historical restoration attempts for seagrass replanting including results, rationale for methods, and related costs;
- Fonseca et al. (1982b) report guidelines for a specific restoration action to transplant eelgrass. This paper provides updated information to Phillips' (1980) guidelines;
- Phillips (1982) presents an overview of seagrass ecosystems and provides a review of specific projects and methods used for eelgrass restoration;
- Thorhaug (1986) provides an overview of historical seagrass restoration efforts, including eelgrass projects, and suggests areas for further research and improvement;
- Thorhaug (1989) reviews seagrass restoration in terms of its ecological and economic benefits to fisheries and aquaculture. Historical seagrass restoration attempts are reviewed;
- Fonseca et al. (1990a) summarize an eelgrass transplanting project and compare results to recently colonized and long-time existing eelgrass habitats;

- Lewis and Phillips (1981) discuss an experimental seagrass restoration project performed in the Florida Keys using various types of planting materials;

- Thorhaug (1981) describes the reliability of several seagrass restoration attempts performed in south Florida, the west Florida coast, the Texas coast, and on the upper Gulf of Mexico;

- Derrenbacker and Lewis (1982) evaluate three methods and Thorhaug (1983) reviews and evaluates the technical feasibility of seagrass planting in an area off Key Largo, Florida, which had been affected by water pipeline installation;

- Durako and Moffler (1984) assess the technical feasibility of seagrass restoration using varied growth mediums and anchoring systems;

- Hoffman et al. (1982) review several historical restoration projects performed in Tampa Bay on affected seagrass communities;

- Fonseca et al. (1987a) evaluate the use of basic ecological data for application to the decisionmaking process when implementing seagrass restoration;

- Fonseca et al. (1987b) report on seagrass transplants that were conducted at sites across a broad geographic area in order to assess seagrass shoot generation and coverage rates under different geographic and environmental conditions;

- Thorhaug (1987) describes four large-scale implementation attempts to restore injured seagrass habitats affected by dredging of an intra-coastal waterway channel and construction activities; and

- Fonseca et al. (1990b) report on experimental research conducted on seagrass restoration in Lassing Park, Florida to create a seagrass habitat on a recently filled navigation basin.

2.2.3.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.7 presents a summary of the technical feasibility for each restoration action. Each action should include a monitoring program.

Exhibit 2.7 Overview of technical feasibility of seagrass restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Efforts**	**Legal and Administrative Factors**
Natural Recovery	Generally feasible	Generallly available	Litte constraint	May need to consider replanting	Coordination of monitoring activities
Replanting	Demonstrated as feasible for *Thalassia* under proper conditions	Appropriate donor sites are required Specialized technical expertise to oversee project is required	Planting may be seasonally limited Herbivory	Replanting due to transplant mortality	Permits may be required for both removal of transplant stock and planting

2.2.3.2.3.1 Natural Recovery

Monitoring of natural recovery is a technically feasible action. The recovery of injured seagrass ecosystems is evaluated in Chapter 3.

2.2.3.2.3.2 Replanting

For restoration of seagrass habitats, transplanting can be performed. This method has been attempted for many years and more information is becoming available as the actions continue to develop. Three primary types of propagules are used for replanting, plugs and turfs, shoots (or bare roots), and seeds.

Plugs and Turfs

A plug contains seagrass blades, roots, rhizomes, and sediment. It is extracted from a natural bed and transported to an excavated hole. Small plugs can also be transferred to peat pots, which are then planted in the sediment. Plugs may be anchored in high energy areas using cement collars that weigh the transplant down or by covering the transplant with chicken wire. The plug transplant minimizes "trauma" to the roots and rhizomes of the seagrass plant because it entails removing a large portion of the sediment mass with the plant. This method provides seagrass plants with immediate sediment stabilization.

A coring mechanism is used to perform the transplant of a seagrass plug. In transplant experiments a PVC coring tube (approximately 10 cm in diameter and 51 cm in length) was used to obtain a seagrass plug from a donor seagrass bed and insert it (with the plug intact) into the receiving sediment. The cored seagrass plugs are installed using a tree-planting bar, which is used to loosen sediment. The coring transplant operation is most efficient when a team of individuals work in the preparation, handling, and insertion of planting materials (using SCUBA gear is necessary if the planting is done below a certain water depth).

The peat pot method is presently being developed. Experiments are currently being conducted in order to evaluate its technical feasibility, cost-effectiveness, and success (Fonseca, 1991). This method uses small seagrass plugs as transplant material. Plugs are placed in square peat pots that help to support the plug and its roots. The potted plug is then planted into the habitat sediment. One advantage of this method is the ability to place fertilizer pellets in the pots to enhance the growth process. The peat pot method is considered a feasible means of transplanting mature seagrass stocks. However, the long-term success of this action remains uncertain (Fonseca, 1991; see Chapter 3).

The use of seagrass turfs (also known as "sods") entails cutting out a piece of sediment from the donor habitat and placing it in a shallow trench cut at the recipient site. Seagrass transplanting using turfs has been demonstrated as technically feasible (Thorhaug and Austin, 1976; Thorhaug, 1980; Phillips, 1982; and Thorhaug, 1986). The use of plugs and turfs is considered to be the most technically feasible approach for eelgrass restoration (Thorhaug and Austin, 1976; Thorhaug, 1986; Phillips, 1982).

Shoots (Bare Root)

Seagrass shoots are bare-root plants collected (for replanting purposes) from donor seagrass beds. The use of seagrass shoots for transplanting often requires anchoring by staples to stabilize the root system within the receiving sediment. It is common to combine several shoots together in order to provide a more complex root base. The logistics of using seagrass shoots are often simpler than handling seagrass plugs due to the lack of sediment associated with shoots. Transplants of seagrass shoots were attempted for many species of seagrasses and are technically feasible for eelgrass (Thorhaug and Austin, 1976; Thorhaug, 1980; Phillips, 1980 1982; Thorhaug, 1986; and Fonseca, 1990).

In the staple method, seagrass "planting units" are made from several seagrass shoots. The planting unit is then inserted into the sediment and stapled by hand with the aid of snorkel or SCUBA equipment. Stapling is more reliable than other shoot actions (Fonseca et al., 1990b). However, experiments using this method have shown greater loss rates in areas exposed to high turbidity and wave action during low tides.

Seeds

Seagrass seeds are also used to recolonize an injured seagrass habitat. Seeds are planted by hand after being gathered from a donor bed by separating the seeds from the fruit pod. Habitat areas with low turbulence are more easily seeded and have a greater chance of root formation and growth. Seeds can grow in either barren sediment, established seagrass beds, or in benthic algae (Thorhaug, 1989). The seeding action requires less labor than transplant actions, and so is potentially cost effective as a seagrass restoration action if abundant seeds are available. However, this method depends largely on the seasonal availability of seeds. There is difficulty in collecting seagrass seeds for replanting and such replanting efforts often result in poor germination rates, particularly for species other than *Thalassia* (Thorhaug and Austin, 1976; Fonseca et al., 1979; Thorhaug, 1980; Phillips, 1982; and Thorhaug, 1986). Thus, it is presently only technically feasible for *Thalassia*.

Availability of Services, Materials, and Equipment

Seagrass plant materials are commonly hand-collected from a "donor" seagrass site using shovels and other tools, depending on the type of plant material to be collected. Technical expertise is required to oversee projects. This would typically involve specialists from academia, government agencies, or firms with experience in seagrass restoration. In addition, divers may be required if restoration is conducted at deeper water depths.

Constraints

Because of variation in growth by season, planting times must be coordinated with the local climatic conditions (Fonseca, 1990a). For example, the fall season is generally considered the best time to plant eelgrass (Fonseca et al. 1979).

Future Restoration Actions

Replanting may be necessary due to transplant mortality.

2.2.3.3 Freshwater Aquatic Beds

Restoration actions consist of:

- Natural Recovery; and
- Replanting

2.2.3.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.3.3.2 Replanting

Replanting of freshwater aquatic beds appears to be technically feasible. However, little documentation exists describing restoration efforts (see Section 3.2.3.3).

2.2.4 Mollusc (Oyster) Reefs

Mollusc reefs include oyster reefs and mussel reefs. Oyster reefs are more prevalent than mussel reefs and support an established fishery. Mussel reefs primarily exist in the more temperate regions and support a less significant fishery. Most previous restoration efforts for mollusc reefs have been for oyster reefs and that is the focus of the discussion here. No literature on mussel reef restoration was identified.

Natural oyster reefs are created when layers of oyster shells cover the substrate, forming a bed. The bottom substrate, or cultch, is commonly a hard smooth surface such as rock bottom or created from deposits of oyster shells. Oyster spat (larvae) attach to the cultch when settling. The oyster reef is formed as the elevation of the bed rises, resulting from the accumulation of dead shells underneath the new spat. Productive oyster reef habitats are generally characterized by cultch mounds that have high elevations and large quantities of exposed surface shells (Morales-Alamo et. al., 1990).

The restoration actions identified in the literature for oyster reef restoration include:

- Natural Recovery;
- Reef Reconstruction; and
- Oyster Reseeding.

These actions can be used for direct restoration or replacement if a suitable site is available.

2.2.4.1 Oil Related Literature

In no cases where oil contamination to oyster habitats was been documented, were direct restoration projects attempted, other than allowing natural recovery to occur (Benefield, 1992; Heil, 1992; Ray, 1992; Soniat, 1992).

2.2.4.2 Non-oil Related Literature

Available literature primarily identifies restoration practices for oyster reefs following natural and human-related adverse influences including hurricanes, siltation, dredging, barge groundings, and other non-oil impacts. The technical feasibility of restoration is addressed in Berrigan (1988a,b, 1990), Bowling (1992a,b), Hofstetter (1981), and Marwitz and Bryan (1990).

2.2.4.3 Technical Feasibility of Restoration Actions

Exhibit 2.8 provides a summary of restoration actions identified for oyster reef restoration. Each action should include a monitoring program.

The following restoration actions are applicable to oyster reefs in both the intertidal and subtidal zones. The literature on oyster reef restoration focuses primarily on reefs in intertidal areas. However, experiments have shown that the success of restoration performed on reefs located both inshore and offshore is not significantly different (Haven et. al., 1987).

Exhibit 2.8 Overview of technical feasibility of mollusc reef (oyster) restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery Monitoring	Generally feasible	Generally available	Little constraint	Reef restoration action could be warranted due to slow recovery	Coordination of monitoring activities
Reef Reconstruction	Demonstrated as feasible	Shell is limited in supply; alternative materials are available	Site selection in the case of off-site restoration	Additional restoration may be necessary if recovery is slow	Coordination with habitat management authorities Permits may be required
Oyster Reseeding	Demonstrated as feasible	Seed oysters may be limited in certain regions	Residual contaminants; poor water quality	Additional seed oysters may be required due to level of mortality	Coordination of activities with habitat management authorities

Exhibit 2.8 (continued)

Vegetation Cropping	Vegetation Cropping		Generally feasible	Readily available	Possibility of further injury	Collateral injury may result in additional restoration	No formal requirements
New Wetland Creation	New Wetland Creation	Soil Removal/ Replacement	Technique has been developed	Variable, since projects may range from simple services to massive construction projects	Acquisition of site Establishment of hydraulic regime Controlling contaminants Pest species	Most viable projects have extensive programs of evaluation and mid-course correction	Permit procedures Negotiation for site acquisition
Low Pressure Flushing	Low Pressure Flushing	Vegetation Cropping	Feasible in limited circumstances	Available from oil spill cleanup contractors	Access to marsh interior	Additional restoration due to damage	No formal requirements
Bioremediation	Bioremedia-tion	New Wetland Creation	Technique is currently being developed	Services and equipment generally available	Few people have strong bioremediation expertise in estuarine and marine systems Work crew access needed Possible eutrophication effects	None expected	Permits required

Several oyster reef restoration actions were performed where the oyster resources are important to the fishing industry (e.g., Maryland, Florida, Gulf of Mexico). These restoration actions proved technically feasible (Berrigan, 1990; Bowling, 1992a,b). The restoration of oyster habitats typically involves:

- Reconstruction of oyster reef substrate using alternative materials; and/or

- Reestablishment of the injured habitat or other comparable site with seed oysters.

2.2.4.3.1 Natural Recovery

Monitoring of natural recovery for oyster reefs is technically feasible in all cases. See Chapter 3 for a discussion of recovery potential.

2.2.4.3.2 Reef Reconstruction

The objective of reef reconstruction is to provide a clean, hard substrate for oyster spat (settled larvae) colonization and growth. The placement of suitable substrate, or cultch, is a action for increased oyster colonization if it is performed in areas with appropriate bottom types (i.e., conducive for immediate oyster set) (Kennedy, 1991; Webster and Meritt, 1988). In general, oysters settle best on bottom that is firm, such as those of rock, stone, or shell. Firm or sticky mud is also a suitable bottom type, but sandy habitats are often subject to shifting, which can result in sedimentation and siltation of the oysters.

Suitable bottom types are often cultivated in oyster producing grounds by laying down a firm substrate "foundation" to support the colonization of oyster spat (Webster and Meritt, 1988). It is common practice among oyster habitat managers to apply cultch in historical oyster producing grounds in order to improve substrate characteristics and increase productivity.

Like bottom type, cultch material must also be of a firm consistency, suitable for larval attachment. Cultch planted in areas where natural oyster reproduction occurs stimulates larval setting and establishment of new oyster populations (Berrigan, 1990). Clean substrate, that which is free of sediment and other organisms, is preferable cultch material for maximum larval attachment.

Availability of Services, Materials, and Equipment

Alternative materials for creating suitable cultch for oyster colonization and growth have been experimented with extensively and include:

- Dredged or Fresh Shell (Oyster or Clam);
- Limestone;
- Cement Compounds;
- Slate and Shale;
- Gravel;
- Tire Chips; and
- Coal Ash.

Shell. Both oyster and clam shells have been used as cultch material. The shell is dredged from areas with large deposits of shell material or from other sources such as oyster processing plants.

For oyster reef restoration projects performed in Maryland, Florida, and Texas, shell was selected for use as the designated cultch material (Maryland Department of Natural Resources (MDNR), 1992; Berrigan, 1990; Hofstetter, 1981a,b; Marwitz and Bryan, 1990; Bowling, 1992a,b). In these projects, shell was considered a superior material because of its ability to form a firm base and attract numerous oyster settlements. It is also preferable because of its greater surface area per unit volume, allowing more space for the settlement of oyster larvae (Ray, 1992). It is recommended that shell be planted as cultch at places where maximum larval sets are expected to occur and at favorable times of the year (Hargis and Haven, 1988).

Although shell is the preferable material for oyster cultch, availability in some regions (e.g., Gulf of Mexico) is limited due to restrictions on dredging activities (Abbe, 1992; Benefield, 1992; Judy, 1992; Heil, 1992; Ray, 1992; Soniat, 1992). Experiments were conducted on other materials. These alternatives, discussed below, are not currently in widespread use, but results of recent experiments conclude that some may be viable alternatives (Soniat et al., 1991; Haywood and Soniat, in press).

Limestone. Limestone was recently tested as a potential cultch (Soniat et al., 1991). Limestone may be a feasible alternative to shell because experiments show that limestone is successful in attracting oyster larvae, most likely due to its calcium carbonate composition, the availability of limestone is not limited, and costs are comparable to or lower than shell.

In soft bottom habitats, limestone is not as cost-effective as shell since limestone has a higher weight per unit of volume, thereby requiring a greater volume of material to compensate for sinkage. Yet in tests comparing limestone and shell where sinkage is not a factor (i.e., in hard bottom habitats), limestone proved the preferred cultch material because of its lower cost per unit of volume (Soniat et al., 1991). However, limestone has not yet been used for a large-scale restoration project (Soniat, 1992; Benefield, 1992). In future restoration projects, it is expected that limestone will prove to be a biologically feasible, cost-effective, and environmentally benign alternative to shell as oyster cultch (Soniat et al., 1991), particularly in areas where shell cultch is limited.

Cement Compounds. Other alternatives to shell for oyster cultch include the use of cement compounds, crushed road bed (concrete with some asphalt), gypsum, and "gypment" (gypsum and cement mixture). In a study that compared the effectiveness of crushed road bed and cement with shell as oyster cultch, the shell attracted more oyster spat than the concrete/asphalt mixture (Soniat et al., 1991). In addition, this material is much heavier than shell and results indicated that the road bed may contain trace pollutants.

In the same study, gypsum (a by-product of fertilizer production) was also tested as an alternative to shell and found to attract oyster larvae. Gypsum is relatively lightweight and inexpensive. However, gypsum was extremely soluble in water, therefore not feasible for cultch. A later experiment tested a stabilized gypsum-cement compound ("gypment") as an alternative cultch (Haywood and Soniat, in press). The rate of dissolution of gypment was observed and its effectiveness compared with limestone and shell. Preliminary results indicate that gypment is suitable as cultch, performing as well as or better than shell in attracting oyster spat. Gypment is also acceptable in material weight (i.e., lighter than limestone) and solubility (the stabilizing cement makes the compound insoluble). Gypment is not yet manufactured or used on a large-scale basis, but should be a viable alternative in the future.

Slate and Shale. The use of slate and shale as alternative cultch material was examined (Haven et al., 1987; Mann et al., 1990). Slate was investigated because of its composition (i.e., it is a hard substrate), low cost, and plentiful supply (Haven et al., 1987). It was found that slate attracted a much lower density of oyster spat than shell. Expanded shale was found less effective for oyster larvae settlement in comparison to shell (Mann et al., 1990). However, shale has potential value as a bottom stabilizer prior to substrate placement. The results of these studies favor the use of shell over both slate and shale as a setting medium, but also note that these materials offer a greater per unit area for spat recruitment than shell.

Gravel. The use of gravel as a substitute for oyster cultch was tested and compared to shell, limestone, and concrete (Soniat et al., 1991). The resulting minimal larval setting indicated that gravel is not a biologically acceptable material and thus not a viable option for oyster cultch.

Tire chips. In a study where tire chips (shredded tire casings) were used as an oyster reef substrate replacement, it was found that tire chips are less effective than shell because of dispersal of tire material by currents and wave action (Mann et al., 1990). Other applications of recycled rubber as cultch material are currently being investigated by oyster reef management teams (Judy, 1992).

Coal Ash. Coal ash, a by-product of coal powered plants, is presently being investigated as cultch. A recent study performed in Texas indicated that coal ash may be an acceptable action in terms of effectiveness, availability, and cost (Ray, 1992; Soniat, 1992). However, this material is not yet used on a large-scale basis.

Expertise on oyster habitat management and restoration is available in state fishery management agencies and the scientific community in the primary oyster regions. Reef reconstruction generally requires marine construction services for the placement of materials (i.e., barges and hoses). These requirements are available in most coastal regions.

Constraints

If replacement of oyster reefs is the chosen alternative, it is important to select sites where spat setting was successful in the past. For successful recruitment of oyster spat, placement of reef substrate should be timed with the seasonal cycle of oyster spat settlement. For example, substrate that is planted too early may be fouled by other organisms or by sedimentation, reducing space for larvae to set. If reef substrate is planted too late in the season, the peak oyster settlement period may be missed.

Future Restoration Actions

Future restoration actions, such as the placement of additional reef materials, may be needed if recovery is slower than expected.

2.2.4.3.3 Oyster Reseeding

The technical feasibility of reseeding oyster beds has been demonstrated by regional oyster management agencies (MDNR, 1992). It is common practice for managers of regional oyster fisheries to cultivate seed oyster grounds for annual restocking purposes. Seed oysters are small, not fully-developed oysters which are raised in hatcheries or specially designated natural oyster beds. The rate of oyster reef restoration may be enhanced by transplanting seed oysters onto the reef site or to an established reef habitat elsewhere.

In documented restoration actions performed for injured oyster reef habitats, restocking the reef with seed oyster was not a priority action. The literature indicated that the primary objective of the restoration projects was to reestablish the habitat through replacement of the substrate. The seeding was demonstrated as technically feasible in areas where natural occurrences and fishing resulted in depleted oyster stocks (Munden, 1974; MDNR, 1992).

Availability of Services, Materials, and Equipment

Seed oyster stock used for reseeding is commonly supplied by neighboring seed beds that are cultivated by independent (commercial) oyster harvesters or regional management agencies. Seed stocks are generally locally available, except in cases where all stocks in the proximity of the injured reef area are destroyed or not ready for cultivation. In these situations, seed stock may be obtained from other regions. Proper equipment and expertise required for obtaining and transporting seed oysters are generally accessible.

Constraints

The use of seed oyster stock to reestablish an injured oyster bed may not be feasible when the injured (i.e., contaminated) habitat has not fully recovered to suitable environmental conditions for growth. For example, residual oil or other contamination in the water may affect the development of oyster stock and cause mortality to the juvenile oysters.

Future Restoration Actions

Further reseeding or new reef creation may be needed if recovery goals are not met.

2.2.5 Coral Reefs

Restoration actions for coral reefs include:

- Natural Recovery;
- Reconstruction of Reef Substrate; and
- Coral Transplants.

Reef restoration may be performed as a direct restoration action or as replacement action if a suitable site is available.

2.2.5.1 Oil Related Literature

Little or no empirical work has been done in the area of restoring oil-injured coral reef habitats (Bright 1991; Gittings, 1991b; Hudson, 1991).

2.2.5.2 Non-oil Related Literature

Restoration actions that are reported in recent academic literature focus primarily on the rehabilitation of reef areas injured as a result of structural injury, such as from ship groundings. The reported restoration approach entails the transplanting of live coral pieces or groups of corals from a donor site to an injured reef area. This method is documented as technically feasible (NOAA, 1991b; Fucik et al., 1984). One reef restoration action that is recommended in the literature for use on oil-injured coral reefs involves the transplanting of coral colonies onto the reef frame (Fucik et al., 1984). However, this method has not been employed in oil-related reef injury situations.

2.2.5.3 Technical Feasibility of Restoration Actions

The technical feasibility of restoration actions is summarized in Exhibit 2.9 and is discussed below. Each action should include a monitoring program.

2.2.5.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.5.3.2 Reconstruction of Reef Substrate

Injured reef substrates may need to be reconstructed or reestablished. For example, impact from a ship grounding may fracture the calcium carbonate substrate that forms the coral reef framework. An approach commonly used to restabilize such injured reefs is the use of a calcium carbonate-based cement to fasten broken pieces of the reef back on to the injured areas. The cement used is of similar chemical makeup to coral and is compatible with reef organisms.

Experimental evidence supports the technical feasibility of this action to restore the reef framework (Hudson, 1991). Live corals can recolonize the injured areas where cement is used to restabilize the habitat. The additional support and relief offered by restabilizing the reef substrate enhances the ability of the coral community to regenerate after injury occurs. Relocation of large dislocated sections, such as coral colonies or "coral blocks," onto the reef structure recreates the complex arrangement of the natural coral reef.

Exhibit 2.9 Overview of technical feasibility of coral reef restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Efforts**	**Legal and Administrative Factors**
Natural Recovery Monitoring	Generally feasible	Generally available	Little constraint	Reconstruction and transplants can be considered if necessary	Coordination of monitoring activities
Reconstruction of Reef Substrate	Demonstrated as feasible for structural injury	Specialized scientific expertise required	Suitable environmental conditions required	Transplants can be considered if necessary	Permits required
Coral Transplants	Demonstrated as feasible for structural injury	Transplant stock may be limited	Proper donor mateiral is required	Additional restoration due to transplant mortality	Permits required

Availability of Services, Materials and Equipment

The calcium-based cement used to stabilize the reef structure is a widely-available product (Hudson, 1991). Divers are required to perform the reconstruction. Such services are generally available in areas where there are coral reefs. Scientific expertise will be needed to oversee the operations.

Constraints

There are few constraints on this action, providing care is taken not to cause additional injury. Suitable environmental conditions are required, including that the site be free of contamination.

Future Restoration Actions

If restoration goals are not met, transplants may be considered.

2.2.5.3.3 Coral Transplants

Reef restoration using live coral colony transplants was suggested for oil-injured coral reefs (Fucik et al., 1984), and demonstrated as technically feasible for reefs injured from structural impact.

Coral transplants were used to rebuild an injured reef in a restoration project described by Hudson and Diaz (1988). A higher rate of mortality was observed for transplanted soft corals than for hard corals because of the difficulty of relocating specimens without incurring injury to delicate holdfast tissue. Transplanting hard corals does not pose this problem due to the protection of the tissue by a stony skeleton (Hudson and Diaz, 1988). Full recovery of injured coral reefs restored by the use of transplants is not documented in the literature, primarily because of the length of time required for full growth and natural recovery of coral specimens. (See Chapter 3 for discussion of recovery.)

Availability of Services, Materials and Equipment

The transplant approach to recolonization involves pruning uninjured live coral from nearby reef structures and fastening them to the injured reef. The availability of coral colonies for transplant material is dependent upon the quality and complexity of existing coral stock in the region where the injury occurred. The material used to fasten coral transplants to the injured reef area is a calcium-based cement, a product widely available (Hudson, 1991).

Past restoration efforts of injured coral reefs documented in the literature represent a collaboration of individual scientific expertise. Such expertise would have to be sought from the scientific community.

Divers are required to hand-carry the coral specimens from the boat to the transplant plot and to cement the transplants in place. Such services are generally available in areas inhabited by coral.

Constraints

A review of past coral reef restoration actions using coral transplants identified several considerations for collecting and transplanting coral specimens and transplanting them to the reef framework. Before specimens are transplanted, the substrate must be prepared so that all loose sediment and rock debris and soft coral skeletons are removed from the area. Corals for transplant stock should be selected from existing reefs so that they represent the density and type of corals injured. Impacts to donor sites must be considered.

Coral species selected for transplant material should include specimens that are abundant and fast growing, have mature growth formations, and can be easily attached to the reef substrate. In addition, it is important that the corals selected for transplanting are those with mature reproductive functions and that sources of opposite gamete type are available within the transplant area (Fucik et al., 1984). These criteria ensure that the establishment of new coral growth occurs as quickly as possible. Technical feasibility is dependent on environmental conditions conducive to growth. For instance, observations from transplant experiments include a high survival rate in areas protected from violent wave action, and a reduced rate of recovery at a site chronically polluted (Fucik et al., 1984).

Future Restoration Actions

Additional restoration may be necessary due to mortality of coral transplants.

2.2.6 Estuarine and Marine Intertidal Habitats

This section discusses rocky shores, cobble-gravel beaches, sand beaches, and mud flats in estuarine and marine intertidal habitats. Many shoreline restoration actions are related to and sometimes considered part of discharge cleanup. Restoration is assumed to occur some time after the discharge incident, typically weeks to months, and may include removing residual contamination within beach sediments or removing residual stains or oiling on hard beach surfaces. Such actions may be properly motivated more by aesthetics or other non-biological values than by facilitating recovery of the intertidal biological community. In these cases, restoration is of non-biological services.

2.2.6.1 Intertidal Rocky Shore

Restoration actions consist of actions to remove residual contamination. This cleaning may be needed in addition to response actions because cleanup was inadequate. While it is generally not possible to remove or replace solid rock substrates, oiling or staining of rocky surfaces can often be cleaned to remove surface traces of material. The relevant actions for rocky intertidal habitat restoration include:

- Natural Recovery;
- Sand Blasting;
- Steam Cleaning;
- Flushing; and
- Bioremediation

Actions discussed for rocky intertidal habitats also apply to manmade structures, such as piers, bulwarks, breakwaters, etc.

2.2.6.1.1 Oil Related Literature

The evaluation of the technical feasibility of restoration alternatives in rocky shore intertidal habitats was conducted using several oil discharge-related literature sources. Owens et al. (1992), Hawkins and Southward (1992), Klokk et al. (1983), Anderson et al. (1983), van Oudenhoven (1983), Jahns et al. (1983), Lehr and Belen (1983), and Owens et al. (1983) discuss natural recovery following oil discharges in intertidal habitats. Literature by the Johnson and Pastorok (1985), Der (1975), and Benyon (1973) were used in the evaluation of sand blasting and steam cleaning, along with interviews with John Whitney of NOAA (Anchorage, AK) and Jacqueline Michel of Research Planning, Inc. (Columbia, NC), Anderson et al. (1983), Howard and Little (1987), and Owens et al. (1992) were detail flushing in intertidal areas. The use of chemical remediation in flushing operations is discussed by Fingas et al. (1991), Owens et al. (1992) and the American Petroleum Institute (1991); Richard Lessard of Exxon was also contacted and interviewed in this analysis. Finally, numerous sources cover the developing practice of bioremediation. Hoff (1992), Pritchard and Costa (1991), Greene (1991), Jones and Greenfield (1991), Lee and Levy (1991), Chianelli et al. (1991), Glaser et al. (1991), Minugh et al. (1983), Tramier and Sirvins (1983), Owens et al. (1992), and the U.S. Environmental Protection Agency (1990) appear in the literature. Interviews were also conducted with relevant bioremediation experts, including Russell Chianelli and James Bragg of Exxon and Alain Drexler and Paul Benn of Elf-Aquitaine.

2.2.6.1.2 Non-oil Related Literature

The literature that discusses restoration in intertidal habitats deals primarily with oil-related contamination. Thus, non-oil literature was not reviewed.

2.2.6.1.3 Technical Feasibility of Restoration Actions

The technical feasibility of each action is summarized in Exhibit 2.10 and discussed below. Note that the findings in Exhibit 2.10 apply equally to lacustrine rocky shore habitats, subsequently presented in Section 2.2.8. Each action should include a monitoring program.

2.2.6.1.3.1 Natural Recovery

Monitoring of natural recovery is a technically feasible action in intertidal rocky shore habitats. See Chapter 3 for a discussion of recovery.

2.2.6.1.3.2 Sand Blasting

In cases where residual staining remains on rocky surfaces following the initial cleaning and weathering of oil, it is technically feasible to use sand blasting to remove the stains. Sand blasting involves scouring the affected surface with an abrasive (e.g., sand) propelled by compressed air. Although sand blasting has had limited use in restoration efforts historically, Der (1975) noted that sand blasting rocks following the 1969 Santa Barbara oil well blowout was the "only treatment found effective" in cleaning the rocky habitat affected.

Sand blasting is expected to cause additional impacts, including the disturbance or mortality of organisms, contamination from unrecovered abrasive or oil, and removal of organisms from the habitat by high pressure jets (Johnson and Pastorok, 1985). Oil freed by sand blasting may combine with the abrasive to form a pavement-like coating on rocky surfaces and unrecovered oily abrasive may be ingested by organisms.

Sand blasting is considered primarily a polishing action in high amenity rock areas (e.g., areas with a great deal of recreational interest). The deployment of sand blasting equipment and personnel is fairly straightforward. Sand blasting crews will be deployed either by boat or land, depending upon access to the contaminated area. Necessary sand blasting equipment can be carried on a boat or transported by land to the affected shoreline. Work crews equipped with hoses then direct the abrasive to the contaminated areas.

Recovery of the loosened oil and abrasive may be problematic. Any freed contaminant that enters the water and floats may be contained by booms and sweeps and recovered with vacuum pumps. Similarly, oil and abrasive freed and remaining on shore may be vacuumed. Abrasive entering the water column, however, will likely not be contained and may present additional contamination problems.

Exhibit 2.10 Overview of technical feasibility of rocky shore restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery	Generally feasible	Generally available	Generally available	May require cleaning of areas contaminated by freed oil	Coordination of monitoring activities
Sand Blasting	Generally feasible	Readily available nationwide	Lethal to biota surviving the oiling Strong wave action may limit operations Recovery of abrasive material Access to site important	Freed oil and/or abrasive may need to be recovered	None expected
Steam Cleaning	Feasible for small areas only	Readily available nationwide	Lethal to biota surviving the oiling Access from shore is needed	Freed oil may contaminate previously clean areas	None expected

Exhibit 2.10 (continued)

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Flushing	Generally feasible	Generally available in coastal areas Chemical restoration agents available	Boats must be able to access site May not always remove all stains Removal of organisms Requires temporary storage site for recovered oil	Possible reoiling if freed oil escapes containment system	Permits may be difficult to obtain for chemical restoration
Bioremediation	Technique is currently being developed	Services and equipment generally available Possible difficulty in obtaining fertilizers	Few people have strong bioremediation expertise Work crew access to shore is critical Possible eutrophication effects	None expected	Thorough documentation of efforts Permits required

Availability of Services, Materials and Equipment

The services, materials, and equipment needed for sand blasting are readily available in all regions of the United States. Sand blasting can be performed by trained construction workers. These skills are readily available nationwide.

Constraints

Sand blasting rocky intertidal habitats in high wave energy environments may present certain operational constraints when the contaminated areas are not accessible from shore. Heavy seas will limit access by boat and may endanger work crews. The recovery of abrasive in a high wave energy area may be all but impossible. In cases where sand blasting must be conducted from the sea, the recovery of freed oil and abrasive may be difficult. This is also true for land-based efforts in which recovery equipment cannot operate.

Future Restoration Actions

Future restoration actions may be required in cases where a "pavement" forms on rocks from unrecovered oil and abrasive or when unrecovered abrasive poses a threat to organisms in the area. Recolonization of intertidal organisms may need to be enhanced. However, technically feasible actions to do so have not been documented to date.

2.2.6.1.3.3 Steam Cleaning

Discussions of steam cleaning are largely absent from recent restoration literature. This action involves using steam applied steadily and slowly by shore-based crews through hoses or some type of jet to loosen weathered oil clinging to rocks. Oil that is loosened by steam flows to lower sites on the shore where it is dissipated by wave action or recovered by work crews (Der, 1975). Steam cleaning is distinct from hot water, high-pressure spraying in that water is heated to boiling (212° F) in steam cleaning, but only to 140° F in hot water spraying (Michel, 1993).

The Johnson and Pastorok (1985) discusses this action, along with its possible impacts, which include the disturbance or mortality of organisms, contamination by unrecovered oil, crushing of organisms by personnel or equipment, disruption of sediments, or re-oiling of surfaces. Der (1975) notes that steam cleaning was used in rip rap areas following the blowout of an oil platform off of Santa Barbara in 1969. Although oil was loosened by the steam, the action left a black coating of oil on rock surfaces.

Steam cleaning is typically used as a polishing action in high amenity rocky areas (e.g., areas with a great deal of recreational interest). Steam cleaning is also appropriate for manmade structures, rip rap, and sea walls (Michel, 1993). Steam cleaning usually must be conducted from shore, since it is typically performed in the upper intertidal zone only (Michel, 1993). This is because steam must be directed at stains steadily, and must be deployed near the stain. All necessary equipment may be carried on a boat or transported by land to the shoreline. Work crews equipped with hoses or jets direct the steam to the contaminated areas.

Recovery of the loosened oil is frequently accomplished by vacuuming oil from water or rocks. If oil is allowed to flow into the sea, it must be contained using booms or sweeps to prevent additional contamination of other areas.

As in the case of sand blasting, steam cleaning is technologically fairly simple. Therefore, few factors influence its technical feasibility.

<u>Availability of Services, Materials and Equipment</u>

Materials and equipment should be readily available nationwide to perform the steam cleaning process.

<u>Constraints</u>

Since operators must be able to direct a steady flow of steam to the weathered oil, steam cleaning is a slow and time-consuming procedure. Therefore, this action is only feasible for small areas. Crews are also likely be required to operate from shore, since they will operate primarily in the upper intertidal zone. Technical feasibility of this method depends on the access to oiled rocks. It is necessary that crews work unimpeded in contaminated areas for extended periods of time to apply this action effectively.

Steam cleaning may also present occupational health and safety constraints, since workers will be operating adjacent to water heated to boiling. Care must be taken and protection provided to prevent burns to workers.

<u>Future Restoration Actions</u>

Except when oil recontaminates previously-cleaned or unoiled areas, future restoration actions is unnecessary. Since steam cleaning is lethal to intertidal organisms, recolonization may need to be enhanced. However, technically-feasible actions to do so have not been documented to date.

2.2.6.1.3.4 Flushing

Flushing in rocky intertidal habitats, also referred to as "spot washing," can include the use of ambient or heated water to remove residual oil coatings from hard substrate. In an intertidal zone, the loosened substance is likely washed into the nearby water body where it is contained and removed (Owens et al., 1992).

Techniques used in the past have included low, medium, and high pressure flushing. This discussion focuses on medium pressure since it has evolved as the preferred method. The factors affecting technical feasibility for other pressures or temperatures would be similar. This section also discusses the relatively recent development of chemical restoration methods that are used in conjunction with flushing.

The use of pressure washing in the field is described in Anderson et al. (1983) and Howard and Little (1987). Anderson et al. describe efforts following the principal cleanup efforts in the *Amoco Cadiz* discharge off France. Once mousse was removed from the area, the cleanup team focused on removing stains from beaches and rock faces. Following attempts with low-pressure and high-pressure flushing, those working on the restoration settled on a medium-pressure flushing (approximately 50 psi) method as the most effective, least expensive, and safest alternative. Howard and Little (1987) studied the cleaning effectiveness and biological impacts of low-pressure flushing of very fine intertidal sediments. They indicate that low-pressure flushing is effective where oil is viscous, less than 10 cm thick, and sediments are relatively firm. Further, sediments must be sufficiently thick to avoid erosion. Although Howard and Little's work was performed in a sandy intertidal zone, their claim that this action is effective on firmer sediments suggests that this method is applicable to a rocky shoreline. Owens et al. (1992) also recommend medium pressure spot washing (at approximately 100 psi) to remove oil coated on solid surfaces, such as boulders and rock.

A variety of spray pressures and water temperatures may be used in flushing. Fingas (1991) differentiates among cold water deluge, cold water wash, and warm water wash. In cold water deluge, large volumes of water are pumped over a contaminated area. Cold water wash directs ambient sea water via fire hoses to oiled areas. Warm water wash involves spraying heated water (i.e., at approximately 60°C) at moderate pressure (i.e., at approximately 100 psi) onto contaminated areas. Using warm water is better for weathered oil that is what is expected in restoration situations. Restoration involving very high pressure spraying is rarely used now due to environmental and worker safety considerations. Hot water, high-pressure sprayers were employed, however, from both boat and shore following the *Exxon Valdez* discharge (Whitney, 1993).

Chemical restoration involves the use of surface washing agents that emulsify oil coated on solid surfaces. This makes it easier to contain and remove the oil (Fingas, 1991). Using a process known as "detergency," chemical restoration agents are sprayed onto the oiled surface a short period before flushing to loosen the oil. Although extensive laboratory testing was conducted on Exxon Corexit 9580 (which is on the U.S. EPA's National Contingency Plan approval list), it has never been used in an actual discharge incident (Lessard, 1992). This chemical was not approved for use in Alaska following the *Exxon Valdez* discharge.

Possible environmental impacts of this action include removal of organisms from the substrate, or recontaminating adjacent intertidal areas (Owens et al., 1992, see Chapter 3).

Flushing uses low- to medium-pressure water streams (i.e., less than 100 psi) to directly wash sediments and to release subsurface sediments through agitation. Heated (60°C) sea water is pumped through hoses, and applied by workers on the beach. Water used in flushing operations may be heated or ambient, but very hot water may injure biota that have survived oiling. Flushing is begun at the top of the oiled area during low tide, and continued downshore toward the water. Containment booms or sorbent sweeps are placed in the water to collect the freed contaminants. Skimmers or vacuum units are then used to recover the oil. This action requires at least one boat with a portable skimmer to collect oil washed into the water and held in containment booms. An additional boat equipped with a pump to deliver the water to the crew on shore may also be used, although pumping actions may also be performed from shore. The size of the crew will vary depending on the degree of contamination and other conditions.

Flushing is moderately reliable. Flushing will likely clear off some of the contaminants clinging to rock faces. However, some deep stains may remain.

Availability of Services, Materials and Equipment

The services required should be available from a number of discharge cleanup companies and cooperatives nationwide. Containment booms and sorbent sweeps are the principal materials required in the flushing operations, which are readily available in coastal areas. Exxon Corexit 9580 is available from the manufacturer in Texas. All equipment should be available in all coastal regions. No complex or unusual equipment is required for this restoration action. Most discharge cleanup companies and discharge cooperatives have experts in-house who are qualified to perform or oversee this action.

Constraints

This action will be constrained if the contaminated shoreline is in a high wave energy environment since crews will not be able to operate from boats and because oil will escape over containment booms. Further, this method is not feasible for shorelines with limited access or without suitable areas for short- or medium-term storage of recovered oil.

Future Restoration Actions

There is a risk of reoiling the shoreline with contaminants freed by the flushing process. If the containment system fails, previously cleaned or unoiled areas may need additional restoration.

2.2.6.1.3.5 Bioremediation

Bioremediation involves the use of fertilizers, surfactants, and/or bacteria to increase the populations of hydrocarbon-degrading microorganisms (Hoff, 1992). Specifically, bioremediation in the intertidal zone may be accomplished by seeding a shoreline with hydrocarbon-degrading microbe, and/or adding nitrogen- and phosphorus-containing fertilizers to enhance degradation. Fertilizers that may be applied can be of three types, soluble inorganic (e.g., agricultural fertilizers), oleophilic (i.e., chemically "sticky") nutrient formulations, or slow-release (i.e., granular) formulations (Hoff, 1992). Microorganisms that exist on shorelines require nitrogen and phosphorus to metabolize the carbon in oil. When the supply of nitrogen and phosphorus is depleted the degradation rate of oil declines (Owens et al., 1992). An increased level of nitrogen and phosphorus may stimulate the microbe population, thereby maintaining a high rate of hydrocarbon degradation. Also, limitation by oxygen and/or temperature may be important.

Oleophilic (literally "oil-loving") agents adhere to oil and increase the surface area of oil droplets exposed to microbes. In addition, the oleophilic agent discussed below, Inipol EAP 22, contains approximately 10 percent surfactants, which may also increase oil breakdown through dispersion (Hoff, 1992).

The use of bioremediation in restoration efforts is discussed in several recent sources including Hoff (1992), Pritchard and Costa (1991), Greene (1991), Jones and Greenfield (1991), Lee and Levy (1991), Chianelli et al. (1991), Glaser et al. (1991), and the U.S. EPA (1990). Although most of these studies were largely conducted following the *Exxon Valdez* discharge in the Prince William Sound, Alaska, the results of studies should apply more generally to restoration efforts in other marine and estuarine habitats. An early bioremediation effort restoring subsurface soils is described in Minugh et al. (1983) and early field experimentation is described in Tramier and Sirvins (1983).

Minugh et al. (1983) report the results of early field experience in bioremediation in a restoration effort involving subsurface soil contamination following the release of gasoline and diesel fuel from a bulk storage facility. In this test, nutrients and diffused air were pumped into contaminated silty soils. Over a nine-month period, 360 pounds of oxygen were added per day, along with 6,000 pounds of ammonium chloride and 3,000 pounds of sodium phosphate.

Pritchard and Costa (1991) considered application strategies, logistical problems, commercial availability, and the need to deliver nutrients to both surface and subsurface sediments in selecting fertilizers for the Alaska Oil Spill Bioremediation Project. The granular fertilizer Customblen was selected for subsurface soils and was spread using a mechanical seed spreader at a concentration of 0.20 lbs/m^2. Inipol EAP 22 was chosen for surface oil since it was the only commercially available oleophilic fertilizer that could meet site-specific requirements relating to ability of the nutrients to remain at the site of microbial activity for sustained periods, and could be produced quickly and in large quantities. Inipol was applied using backpack sprayers at a rate of 0.1 gallons/m^2.

Chianelli et al. (1991) describe the use of fertilizers in a field test of bioremediation efforts in response to the *Exxon Valdez* discharge. This effort consisted of adding nutrients (i.e., oleophilic fertilizers) only to oiled locations. The authors recommend an application rate for granular fertilizer (e.g., Customblen) of 0.07 lbs/m^2 when no surface oil is seen but subsurface oil is present. The application of granular fertilizer may be achieved using a hand spreader for subsurface oil. They further recommend liquid fertilizer (e.g., Inipol EAP 22) be applied onto surface coatings of oil at a rate of approximately 0.08 gallons/m^2.

Owens et al. (1992) recommend using nutrient-addition bioremediation as a "polishing action" following initial cleanup or when oiling is light and near the surface. They suggest the use of Inipol and Customblen as well. Their recommended implementation of this action is to deploy workers onto the contaminated shore with a small landing craft. Inipol would be applied using airless paint spraying equipment located on the boat, with workers using long hoses for full access to the shore. Inipol must be heated to 32°C. Customblen may be spread using a hand-cranked lawn spreader. They further recommend fertilizer application every two to four weeks to replace nutrients washed away by the tides. While these actions were developed following the *Exxon Valdez* discharge (where access from shore was limited), the actions described may be carried out entirely from shore.

Jones and Greenfield (1991) describe an intensive field effort in the bioremediation of terrestrial soil following a discharge of No. 6 fuel oil from a Florida power plant. This effort, conducted over 194 days, included site alterations to control drainage, the application of nutrients, water, and bacteria, and sediment tilling (to increase aeration). An area of approximately 4,089 m^2 was treated.

Hoff (1992) summarizes the use of bioremediation in several discharges by the Hazardous Materials Response and Assessment Division of the National Oceanic and Atmospheric Administration (NOAA). Hoff reports on the use of Inipol and Customblen following the *Exxon Valdez* discharge and also the application of Customblen alone following a pipeline break and discharge at the Exxon Bayway refinery in New Jersey. In this latter experiment, Customblen was placed in shallow trenches in an area with existing high levels of nutrients. Hoff also reports on microbe seeding efforts as well as open-water bioremediation. Following the collision of three Apex barges with a tanker in Galveston Bay in 1990, the microbial bioremediation product Alpha

BioSea was applied to oiled marsh in which mechanical recovery was determined to be infeasible. Application was made via high-pressure hose from a small boat. Following a well blowout in 1990 that oiled marsh grasses in the Seal Beach National Wildlife Refuge, the microbial product INOC 8162 was hand sprayed along with the commercial fertilizer MiracleGro 30-6-6. Finally, Hoff reports the experimental application of an unnamed microbial product from a Coast Guard vessel following the *Mega Borg* discharge and fire in the Gulf of Mexico. These case histories demonstrate the technical feasibility of a variety of actions. Effectiveness is evaluated in Chapter 3.

Bioremediation is fairly simple to conduct. In essence, it is very similar to fertilizing in landscape work with either a backpack sprayer (i.e., for liquids) or lawn or hand spreader (i.e., for granular nutrients). Items necessary for performing this activity are vehicles (i.e., for access, transportation, and storage of agent), workers, and backpack sprayers and/or fertilizer spreaders. If bioremediation is conducted from the sea, boats will be needed as well.

Availability of Services, Materials and Equipment

The availability of services should not be a problem. At this point, only a few people have a full understanding of this technology (Merski, 1992). Experts in the field seem eager to become involved in additional bioremediation efforts. As additional scientific work is published, expertise will spread. The equipment needed for the deployment of these fertilizing agents is identical to that used for common lawn care. Therefore, there should be no difficulty in obtaining the proper equipment.

Constraints

The operational constraints of bioremediation depend on conditions at the site. Typical operational constraints are related to access to shore and amount of wave energy. Waves that are too strong (and so remove the fertilizer) or too weak (and so too little flushing and oxygen replenishment) will render bioremediation less effective. (See Chapter 3 for more on effectiveness and success.)

Obtaining information on the availability and use of the bioremediation agents can be difficult and time-consuming due to the fact that bioremediation is an evolving technology. Standard guidelines for application are not consistently developed and documented. Developing an application plan for a specific situation may involve considerable communication with experts.

There appear to be few logistical constraints for applying bioremediation agents. In general, this technology is best adapted for light oiling of fine- to medium-grained beaches in moderate wave energy environments (where tidal action will disperse nutrients over an area without washing them away). The necessary equipment is mobile enough for access to many shorelines. Nutrients can also be sprayed or spread from boats or by shore-based crews.

The addition of nutrients may cause concern over eutrophication. This needs to be evaluated for the site being considered. Toxicity of the bioremediation agent must also be assessed.

2.2.6.2 Intertidal Cobble-Gravel Beaches

The restoration actions relevant for the restoration of cobble-gravel intertidal habitats include:

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation; and
- Bioremediation.

2.2.6.2.1 Oil Related Literature

The oil discharge-related literature used to evaluate technical feasibility of restoration actions in cobble-gravel intertidal habitats is summarized here. Anderson et al. (1983), Klokk et al. (1983), Lehr and Balen (1983), Owens et al. (1983), van Oudenhoven (1983), Jahns et al. (1991), Little and Little (1991), and Owens et al. (1992) discuss natural recovery following oil discharges in intertidal habitats. Anderson et al. (1983), Howard and Little (1987), and Owens et al. (1992) detail flushing in intertidal areas. Flushing following chemical restoration is discussed by the American Petroleum Institute (1991), Fingas (1991), and Owens et al. (1992). Richard Lessard of Exxon was also contacted and interviewed for this analysis. Sediment washing is described by Gumtz (1972), Morris et al. (1985), Bocard et al. (1987), and Huet et al. (1989). Sediment agitation in intertidal zones is discussed by Morris et al. (1985), Levine (1987), Miller (1987), Blaylock and Houghton (1989), and Owens et al. (1992). Robert Levine of Arco Marine was also contacted for further information on sediment agitation during this effort. Finally, numerous sources cover the developing practice of bioremediation. Minugh et al. (1983), Tramier and Sirvins (1983), the U.S. Environmental Protection Agency (1990), Chianelli et al. (1991), Glaser et al. (1991), Greene (1991), Jones and Greenfield (1991), Lee and Levy (1991), Pritchard and Costa (1991), and Owens et al. (1992) appear in the literature. Interviews were also conducted with relevant bioremediation experts, including Russell Chianelli and James Bragg of Exxon and Alain Drexler and Paul Benn of Elf-Aquitaine.

2.2.6.2.2 Non-oil Literature

Most of the literature on restoration in cobble-gravel intertidal habitats concerns the restoration of oil-related injury. Thus, non-oil related literature was not reviewed.

2.2.6.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.11 summarizes the technical feasibility of the restoration actions for cobble-gravel intertidal habitats. Note that the findings of this exhibit apply to lacustrine cobble-gravel shore habitats, subsequently presented in Section 2.2.8. Each action should include a monitoring program.

2.2.6.2.3.1 Natural Recovery

Monitoring of natural recovery is a technically feasible option. See Chapter 3 for a discussion of recovery.

2.2.6.2.3.2 Flushing

The use of low-pressure flushing to remove oil adhering to surface materials in sand or gravel beaches and flush it back into the water is discussed in the response literature by Owens et al. (1992) and Howard and Little (1987). Its use has not been documented as a restoration action, but it is technically feasible as a restoration action. Howard and Little discuss the results of field tests of ambient seawater flushing on fine-grained intertidal sands. Water was hosed at a rate of two liters per second toward the lower end of test plots. A side-to-side motion was used to loosen contaminants. Refloated oil was contained using booms deployed in the water, and the oil was collected from the water surface using hand scoops and placed into an oil/water separator. Howard and Little found this action to be very effective in recovering oil from fine sandy sediments. On average, 85 percent of applied fuel oil was recovered. They note that this action works best on relatively firm sediments, but make the important observation that it may be unsuccessful on very coarse sands and gravel due to erosion and the mixing of sediments and oil. Greater permeability of sediments or depth of the water table may impede flushing. Since this action relies on raising the water table, oil and sediment may become mixed when this does not occur.

Owens et al. (1992) recommend the flushing action for fine- and coarse-grained gravel shorelines. The larger sediments on cobble-gravel beaches present an appropriate habitat for flushing. In their recommended action, oil is washed off of sediments and flushed downshore for collection from the water surface. They indicate that highly weathered oil is likely to be somewhat resistant to this action, but that it is ideal for mobile oil and oil coating surface sediments lightly.

Exhibit 2.11 Overview of Technical Feasibility of Cobble Gravel Shore Restoration

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery	Generally feasible	Generally available	Little constraint	May require cleaning of areas contaminated by freed oil	Coordination of monitoring activities
Flushing	Generally feasible	Generally available in coastal areas Chemical restoration agents available	Boats must be able to access site Removal of organisms Requires temporary storage site for recovered oil May drive oil deeper into sediments	Possible reoiling if freed oil escapes containment system May drive oil deeper into sediments requiring further action	Permits may be difficult to obtain for use of chemical agents
Sediment Washing	Generally feasible	Purpose-built equipment not widely available, but can be assembled from available components	Qualified engineer recommended for washer assembly Backshore site required Lethal to organisms	Possible if recolonization of sediments does not occur	None expected

Exhibit 2.11 (continued)

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Sediment Agitation	Technique has been developed	"Muck Monster" technology is patented; must work through Arco Marine Equipment rental may be difficult	Qualified engineer needed to assemble equipment Access to shore by heavy equipment needed Worker safety issues	None expected	Permits required
Bioremediation	Technique is currently being developed, most successful in case of *Exxon Valdez*	Services and equipment generally available	Few people have strong bioremediation expertise in estuarine and marine systems Work crew access to shore is critical Possible eutrophication effects	None expected	Thorough documentation of efforts Permits required

The Corpus Christi Area Oil Spill Control Association (CCAOSCA) utilizes a low-pressure flushing action in most of its discharge responses (Christian, 1991). They have found this to be the least expensive action applicable to a number of habitats, including sandy beaches and erosional scarps. Factors complicating flushing efficiency and cost include shore type and contaminant type. Rubble shores require the most effort of the shorelines to which the CCAOSCA must respond and thicker oil may require repeated flushing to be cleaned.

Potential impacts from flushing include removal or mortality of organisms, habitat disruption, and oiling of clean sediments by freed oil (Owens et al., 1992; see Chapter 3). Flushing actions (including the use of chemical restoration) are described for rocky intertidal habitats in Section 2.2.6.1.3.4. Issues related to technical feasibility in cobble-gravel intertidal habitats are similar to those described in that Section.

2.2.6.2.3.3 Sediment Washing

Bulk oil deposited on gravel, cobble, or sandy beaches is generally removed during cleanup operations either through natural processes or the use of methods such as flushing, vacuum pumping, etc. Once bulk oil is removed from beaches, a residual amount can be found deposited in the substrate. Thus, restoration in these habitats may focus on remediating the contaminated beach materials. Several methods for removing residual contamination have been demonstrated. These include washing the material on site, agitating and flushing the upper layers of material, or depositing the material to the surf zone for natural washing (Johnson and Pastorok, 1985). For this effort, sediment washing involves the containment and removal of contaminants by collecting, washing on-site, and re-distributing beach material.

Several sources discuss the use of sediment washing in field experimentation (Gumtz, 1972; Bocard et al., 1987, Huet et al., 1989; and Morris et al., 1985). Gumtz (1972) details the development and field testing of a mobile beach cleaning (sediment washing) device constructed for the U.S. Environmental Protection Agency. This machine was constructed on a 40-foot trailer and is comprised of a froth flotation machine with a belt feeder for sand, a submersible water pump, an air supercharger, and a diesel electricity generator. After extensive field testing, Gumtz concluded that this mobile cleaner could operate at a capacity of 30 tons per hour.

Bocard et al. (1987) describe tests of a prototype mobile sand-washing plant designed for deployment in the event of oil discharges. Huet et al. (1989) describe the use of such a sand-washing device for cleaning oil-contaminated pebbles. With this technology, contaminated gravel is stripped from the beach and placed into a loading funnel from which it drops into a rotating washing cylinder (i.e., drum scrubber). The gravel is washed with warm water to which a cleaning agent has been added. The gravel is then transferred to a hydro-cyclone to separate it from the wash water. Decanting tanks are used to separate oil from the wash water. After washing, the gravel can be re-deployed on the beach. The throughput of the washing apparatus was demonstrated to be approximately 18 metric tons per hour. Soil with an initial oil content of roughly 5 to 10 percent had an average residual oil content of 0.2 percent after cleaning (for moderately weathered oil).

Morris et al. (1985) report on tests of a device similar to the one described by Brocard et al. (1987). The equipment used was essentially a standard sand and gravel washing plant typically found in quarries, modified to handle a courser feed with an added device to separate the oil from the washing water. The wash water had kerosene added to expedite cleaning. A device was tested that had a throughput of 14 metric tons of beach material per hour. Beach material with an initial oil content of two to six percent was washed to a final product containing 0.15 percent oil.

Anderson et al. (1983) describe a modified use of sediment washing (actually, relocation of oiled sediments to the surf zone). They recommend such an action for low priority, low amenity beaches.

Owens et al. (1992) indicate that sediment washing units may be specially-constructed "drum types or adapted commercially-available equipment, such as portable or truck mounted cement mixers." Further, they indicate that the solutions used for washing may include either water (hot or cold) and/or dispersants or beach cleaning agents. They note that cleaning time is related to the oil type, degree of weathering, loading, and temperature of wash water. Higher wash water temperatures and the use of cleaning agents are likely to decrease wash time. They recommend that wash cycles should begin at 10 to 15 minutes and adjusted to reflect the efficacy of the sediment washing process.

Sediment washing is best suited for use on moderate to heavily-oiled shorelines, especially in sheltered, low energy areas. It is best if the shore is comprised of medium-grained sediments.

Sediment washing will injure organisms (as discussed in Chapter 3). In addition, manual removal of sediment may cause oil to be mixed into the substrate by personnel and vehicle traffic at the site. Substrate removal on cobble shorelines may cause erosion or flooding of backshore areas, erosion of adjacent shorelines, and depletion of offshore sediment deposits. Owens et al. (1992) list some net loss of material and the temporary destabilization of the beach as the potential impacts of sediment washing.

A potential restoration action for these intertidal habitats is a method employing a sand-washing action such as those described above. Sediment washing is conceptually a fairly simple operation. Oiled sediments are removed manually (e.g., with a shovel) or mechanically (e.g., with a front loader), transported to a backshore area, and run through the washing equipment. Washing may be performed with or without detergents or dispersants, depending on the extent of oiling, regulatory requirements, etc. After washing, cleaned sediments are redeployed on the beach (ideally the identical spot from which they were removed). Wash water and recovered oil may be separated, and the wash water reused. Decanted oil is then stored for disposal or transport to a recycling facility.

Availability of Services, Materials and Equipment

The detergents or dispersants used with the sediment washer should be readily available commercially. While "purpose-built" washers are not common, a sediment washer can be assembled from readily-available equipment. Mechanical removal devices (e.g., front-end loaders, backhoes, bulldozers, manual labor, hand tools), conveyors, and cement mixers should be readily available in all regions of the United States.

A qualified civil engineer with road or construction site preparation experience is likely to be able to assemble a beach cleaning apparatus using components from the excavation industry. This assumes the engineer has access to the relevant literature describing sediment washing devices.

Constraints

This procedure requires access to oiled beaches for crews and equipment (e.g., front end loaders, etc.). This method also requires sufficient room in backshore areas for the sediment washer and other equipment. After they have been washed, the sediments should be returned to the exact location from which they were found.

Future Restoration Actions

Future restoration actions may be required if recolonization of sediments by organisms does not occur naturally after the washing process. It may be necessary to begin the recolonization process by transplanting some sediment-dwelling organisms to the cleaned sediments.

2.2.6.2.3.4 Sediment Agitation

Sediment agitation is performed by turning oiled sediments to break up oiled layers and enhance natural degradation processes (e.g., physical, microbial, and photochemical) (Owens et al., 1992). Sediment agitation also allows access to and treatment of subsurface oils.

The agitation of sediments for the removal of stranded oil and emulsion is discussed by Morris et al. (1985), Levine (1987), Miller (1987), Blaylock and Houghton (1989), and Owens et al. (1992). Owens et al. (1992) describe the process, which they call sediment tilling. This involves using a tractor fitted with tines or ripper blades to till sediments near the surface in oiled areas. Morris et al. (1985) report field experiments of various sediment agitation actions for removing water-in-oil emulsions from firm sandy beaches. A variety of equipment configurations were used, including standard vehicle-mounted snowplows, tracked bulldozers, diggers fitted with rubber blades, tractor-mounted scrapers, and front-loader tractors. They found rubber-bladed equipment attached to a front-end loader the best-suited configuration for firm, sandy sediments. Manual tilling is also an option for smaller areas, with the advantage of causing less operational impact.

Levine (1987) and Miller (1987) describe the use of a beach agitation device in the cleanup of the *Arco Anchorage* discharge in Port Angeles Harbor, Washington. A beach agitation device was used in conjunction with high-pressure flushing and vacuum pumping to remove subsurface oil and restore a beach composed of sand, gravel, and cobble.

Arco Marine's patent for the Muck Monster includes modifications to the basic design described in Levine (1987) to replace the bulldozers with log skidders (Levine, 1992). Log skidders are similar to bull dozers, but have large, balloon-type wheels rather than tracks. Although the bulldozers used in the *Arco Anchorage* cleanup used wider-than-conventional tracks, the design for the Muck Monster was modified to use a vehicle with tires to minimize ecological impacts by reducing the load delivered to the sediments and to provide higher ground clearance and greater mobility.

A potential impact from sediment agitation is the mixing of oil deeper into sediments (Owens et al., 1992). Even when beach cleaning machines result in few physical impacts to the beach structure, they may clean only the surface of the beach (Johnson and Pastorok, 1985).

The actions utilized for sediment agitation are assumed to be similar to those developed by Arco Marine, Inc. for the cleanup of the *Arco Anchorage* discharge (Levine, 1987). These actions have been patented by Arco Marine, so it would be necessary for any discharge responder to contact Arco Marine for guidance before using a Muck Monster III for cleanup. Arco Marine assisted in the cleanup of Huntington Beach, California, following the *American Trader* discharge, and did not seek any compensation for its activities beyond seeking reimbursement for cellular phone usage (Levine, 1992).

Evaluation of potential impacts on biota is in Chapter 3.

<u>Availability of Services, Materials, and Equipment</u>

Since the Muck Monster restoration method is patented by Arco Marine, Inc., the trustees involved in restoration must contact Arco Marine to use this method. Although this is the case, there are no anticipated impacts to technical feasibility from this requirement. In cooperation with any discharge response effort, Arco Marine will send personnel to a discharge site at little or no cost (Levine, 1992).

The materials used in carrying out this restoration action (e.g., sorbents, sweeps, etc.) should be readily available in all coastal areas of the United States.

In some cases, there may be some difficulty finding equipment to rent. The bulldozers used in *Arco Anchorage* restoration were effectively destroyed, bearings were degreased, and the generator and electrical system was damaged. An equipment supplier aware that heavy equipment is to be used in salt water may be hesitant to rent it out. Furthermore, log skidders equipment found in timber areas may be difficult to find in some regions (Levine, 1992).

Levine (1992) recommends that a qualified engineer (e.g., from the equipment manufacturer, such as John Deere) be on scene to assist in assembly of required equipment.

Constraints

No construction constraints are expected. The constraints related to the assembly of the Muck Monster are related to expertise and the availability of equipment. Further operational constraints are discussed below.

In cases where access to shoreline is possible using landing craft only, and where tides are high, this restoration method may not be feasible since equipment would need to be moved nightly (Levine, 1992).

Levine (1992) noted that there were additional concerns for occupational health and safety at the Muck Monster job site since workers must work in water. Cold water may increase rate of fatigue. Workers must also wear life jackets and other OSHA-required gear. During the *Arco Anchorage* discharge, workers occasionally fell into the water. There was additional risk to workers laboring near moving heavy machinery.

Finally, the bulldozer operator must pay strict attention to the slope of the shoreline since the Muck Monster is operating in water. Sudden dropoffs and submerged objects present additional hazards.

This action should be limited to mid- and upper-intertidal zones to limit the impact on biota. Further discussion of impacts is in Chapter 3.

Future Restoration Actions

While long-term monitoring may be recommended, additional restoration efforts are not needed. The state of Washington's monitoring requirement was reduced since cleaning was found highly effective (Levine, 1992). Blaylock and Houghton (1989) report the results of a 30-month infaunal sampling project that showed a statistically significant increase in average biomass, density, and species diversity in areas oiled heavily in the *Arco Anchorage* discharge. Similar increases were not shown in unoiled control areas. This indicates that this restoration action is effective without continuing restoration efforts.

2.2.6.2.3.5 Bioremediation

Bioremediation is described in detail in Section 2.2.6.1.3.5. The details and considerations presented in that section also apply to cobble-gravel intertidal habitats.

2.2.6.3 Intertidal Sand Beaches

A review of restoration-related literature indicates that the following restoration actions are applicable to sand intertidal habitats:

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation;
- Bioremediation; and
- Incineration.

This section describes each of the relevant restoration actions and their technical feasibility within the sand beach intertidal habitat.

2.2.6.3.1 Oil Related Literature

Oil related literature for flushing, sediment washing, sediment agitation, and bioremediation is discussed in Section 2.2.6.1.1. Incineration of contaminated sand is discussed by van Oudenhoven (1983) and Eidam et al. (1975).

2.2.6.3.2 Non-oil Related Literature

For the most part, the literature encountered in this effort dealt with the contamination of sand intertidal habitats by oil. However, data provided by Garbaciak (1992) related to the incineration and disposal of sand contaminated by toxic substances. Sand dune restoration in general is discussed by Knudson (1980) and Salmon et al. (1982).

2.2.6.3.3 Technical Feasibility of Restoration Actions

Since the restoration methods for sand intertidal habitats are similar to those for other intertidal habitats, the following sections heavily reference these other sections and do not repeat the findings. The exception to this is the discussion of incineration, which is appropriate for sand sediments only. Exhibit 2.12 summarizes the technical feasibility of restoration actions. Note the findings in this exhibit apply to riverine and lacustrine sandy shore habitats, discussed in Section 2.2.8.3. Each action should include a monitoring program.

2.2.6.3.3.1 Natural Recovery

Monitoring natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.6.3.3.2 Flushing

This action is discussed in detail in Section 2.2.6.1.3.2.

2.2.6.3.3.3 Sediment Washing

Sediment washing in sand intertidal habitats is essentially the same as in cobble-gravel environments. Therefore, see Section 2.2.6.2.3.3. for a discussion of this method.

2.2.6.3.3.4 Sediment Agitation

Sediment agitation is similar in sand and cobble-gravel intertidal habitats. See Section 2.2.6.2.3.4. for a detailed overview of this method.

2.2.6.3.3.5 Bioremediation

Refer to Section 2.2.6.1.3.5. for a detailed description of bioremediation.

2.2.6.3.3.6 Incineration

While incineration destroys remaining biota in the sediments, it is a potential action for restoring services of a sand beach. Most of the literature related to incineration or burning in oil discharge response or restoration discusses the burning of oil slicks at sea or in broken ice. Owens et al. (1992), Buist (1987), Smith and Diaz (1987), Whittaker (1987), Tennyson (1991), Allen (1991), and Evans et al. (1991) discuss burning of oil in these habitats. These are clearly response actions and not restoration and will not be discussed further.

However, there is little discussion in the literature of the incineration of oiled sediments in the intertidal zone, which might be considered a restoration action. Following the contamination of sand sediments in Qatar, several incineration disposal methods were examined for the disposal of contaminated sand (van Oudenhoven, 1983). Combustion was attempted using oil alone, gasoline, kerosene, and a combination of kerosene and driftwood. Following the *Tamano* discharge in the Casco Bay in Maine, incineration and the recycling of sand was considered (Eidam et al., 1975). No appropriate incinerator was found in the New England area which could handle the sand.

Exhibit 2.12 Overview of technical feasibility of sand shore restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery	Generally feasible	Generally available	Little constraint	May require cleaning of areas contaminated by freed oil	Coordination of monitoring activities
Flushing	Generally feasible	Generally available in coastal areas Chemical restoration agents available	Boats must be able to access site Requires temporary storage site for recovered oil Removal of organisms	Possible reoiling if freed oil escapes containment system	Permits may be difficult to obtain for chemical restoration
Sediment Washing	Generally feasible for areas of low ecological sensitivity	Purpose-built equipment not widely available, but can be assembled from available components	Qualified engineer recommended for washer assembly Backshore site required Lethal to organisms	Possible if recolonization of sediments does not occur	None expected

Exhibit 2.12 (continued)

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Sediment Agitation	Technique has been developed	"Muck Monster" technology is patented; must work through Arco Marine Equipment rental may be difficult	Qualified engineer needed to assemble equipment Access to shoreline by heavy equipment needed Worker safety issues	None expected	Permits required
Bioremediation	Technique is currently being developed	Services and equipment generally available	Few people have strong bioremediation expertise in estuarine and marine systems Work crew access to shore is critical Possible eutrophication effects	None expected	Thorough documentation of efforts Permits required
Incineration	Technology has been developed	Mobile incinerators may not be available	Equipment must be able to access site Lethal to organisms Smoke generated must not impact wildlife or humans	Sediment replacement may be required	Permits likely to be required
Removal and Replacement	Generally feasible	Upland disposal site required	Experts must verify removal is required Removal of organisms Sand causeway to site may reduce impacts	Possible if recolonization of sediments does not occur	Likely to be required

In the event that incineration is used to burn contaminated intertidal sand, if available, some type of mobile incinerator could be used. Such an incinerator would be placed in a backshore area adjacent to the area of contamination. Contaminated sediments would be removed manually or by using mechanical equipment. Whether or not mechanical equipment is used would depend on whether the sediments can support their weight and on environmental sensitivity. Removed sediments would then be transported to the backshore area. Debris would be separated and the sand would be fed into the incinerator to burn the oil.

Availability of Services, Materials and Equipment

Services and materials needed for incineration are available in selected locations. The availability of mobile incinerators, however, is an issue that may affect the technical feasibility of incineration. As Eidam et al. (1975) found, incinerators able to handle sand could not be found in New England. It is possible that portable incinerators may not be available in all regions of the country.

Constraints

The use of incineration in restoration assumes that mechanical equipment can be brought onto the beach for sand removal in some cases. If sand cannot support heavy equipment, or if organisms in the area are sensitive, it may be necessary to utilize manual removal of sediments.

Further, it is assumed that there is adequate room in backshore areas for an incinerator. Finally, the smoke generated by incinerator must not adversely affect workers, wildlife, or nearby residents.

Future Restoration Actions

In some cases, removed sediments will be replaced. The removal of sediments from some areas may lead to accelerated erosion. Sediment replacement will naturally increase effort and cost.

2.2.6.4 Intertidal Mud Flats

Mud flats are compacted, fine-grained sediments often backed by sandy beaches or marshes. Mud flat intertidal habitats occur in areas in which general circulation results in sediment deposition (Johnson and Pastorok, 1985).

Unlike the soils found in sand and cobble-gravel intertidal habitats, few technologies have been effectively demonstrated for restoring mud. Sediment washing, for example, requires larger-grained beach material. Flushing is unlikely to be feasible in fluid muds due to sediment-oil mixing (Howard and Little, 1987). Some logistical barriers also exist for using many of the restoration technologies requiring heavy equipment on site. Mud sediments cannot physically support heavy machinery and, thus, access from land may not be possible.

An assessment of restoration literature indicates that the following restoration actions may be relevant to mud flat intertidal habitats:

- Natural Recovery;
- Sediment Removal and Replacement; and
- Bioremediation.

This section discusses the technical feasibility of each restoration action within the mud flat intertidal habitat.

2.2.6.4.1 Oil Related Literature

The literature covering restoration of mud flats deals mostly with oil discharge contamination. Several oil discharge related documents were reviewed. Johnson and Pastorok (1985), van Oudenhoven (1983), and Lehr and Balen (1983) discuss removal and replacement efforts in mud flat discharge remediation. Finally, the same bioremediation sources listed above for other intertidal habitats also apply to mud flat bioremediation.

2.2.6.4.2 Non-oil Related Literature

No literature discussing restoration of mud flats in a non-oil context were identified.

2.2.6.4.3 Technical Feasibility of Restoration Actions

The technical feasibility of each restoration action in mud flat intertidal habitats is discussed below and summarized in Exhibit 2.13. Note that these actions also apply to riverine and lacustrine silt-mud shores, subsequently presented in Section 2.2.8.4. Each action should include a monitoring program.

2.2.6.4.3.1 Natural Recovery

Monitoring of natural recovery is a technically feasible action. See Chapter 3 for a discussion of recovery.

Exhibit 2.13 Overview of technical feasibility of mud flat restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Efforts	Legal and Administrative Factors
Natural Recovery	Generally feasible	Generally available	Little constraint	May require cleaning of areas contaminated by freed oil	Coordination of monitoring available
Sediment Removal and Replacement	Generally feasible	Upland disposal site required	Experts must verify removal is required Removal of organisms Sand causeway to site may reduce impacts	Possible if recolonization of sediments does not occur	Permits likely to be required
Bioremediation	Technique is currently being developed	Services and equipment generally available	Few people have strong bioremediation expertise Work crew access to shore is critical Possible eutrophication effects	None expected	Thorough documentation of efforts Permits required

2.2.6.4.3.2 Sediment Removal/Replacement

Johnson and Pastorok (1985) indicate that manual sediment removal is a viable action for the cleanup of oil discharges in tidal flat habitats. However, residual oil contamination may become mixed into the sediments (Johnson and Pastorok, 1985). This will leave only some type of sediment cleaning or replacement as feasible actions. Since field applications of the sediment washing technologies described for cobble-gravel and sandy intertidal habitats above (Sections 2.2.6.2 and 2.2.6.3, respectively) have not been applied to the finer sediments found in mud flats, the only reasonable manual removal restoration action for the mud flat intertidal habitat is the mechanical removal, disposal, and replacement of contaminated sediments. Van Oudenhoven (1983) reviews restoration efforts in mud flats. He notes that oil on mud flats that was not removed remained there nearly three years after the discharge came ashore. Furthermore, he recommends the use of manual labor on mud flats to minimize ecological injury. A action used in the response involved construction of temporary sand causeways on the flats, manual scraping of oil, transportation of contaminated soil by wheelbarrow to front-end loaders located on the causeway, and placement of removed mud in dump trucks in backshore areas.

Removal and replacement of mud sediments involves removing contaminated soil, loading and transporting it for disposal, and obtaining and deploying replacement soil in its place. This method should use a removal action similar to that described in van Oudenhoven (1983). Work crews remove contaminated mud manually, and transport it (e.g., using a wheelbarrow) to a backshore area. Contaminated soil is piled on site in the backshore area to await loading onto a dump truck for transportation to a disposal facility. Soils containing non-hazardous contaminants will likely not need treatment or stabilization prior to disposal in an upland landfill. Replacement soil is then trucked to the backshore area, transported manually onto the restoration site, and manually spread by workers to a rough finish grade.

<u>Availability of Services, Materials, and Equipment</u>

The services required for this restoration action should be readily available in all areas of the country. Necessary services include basic landscaping labor and trucking. At this point, upland disposal is generally available within a reasonable distance in all regions of the country. Should some type of toxic or hazardous contaminant be involved, however, disposal alternatives will be limited since contaminated soils will need to be transported to a qualified facility meeting Resource Conservation and Recovery Act (RCRA) standards. Cost and logistical problems will increase in these cases.

No restraints to technical feasibility are expected from factors related to materials. The only material requirement for this action is the soil needed for replacing the soil removed. Suitable soil should be available nationwide. Finally, no exceptional equipment needs are anticipated for this restoration action.

Experts qualified to assess whether the drastic measures of removal and replacement should be performed should be consulted before this action is used. Since heavy mortalities to sediment-dwelling organisms are likely in these ecologically-important areas, it should be determined that removal and replacement will have a net benefit to the mud flat habitat.

Constraints

The construction of a sand causeway as described in van Oudenhoven (1983) may be recommended. While this is an additional task in the restoration process, it does not present a significant increase in the level of effort required. Care must be taken when operating in a mud flat habitat to minimize contact with mud sediments. This is recommended for equipment, machinery, and work crews. A sand causeway may be constructed to establish a regular path to work areas and limit traffic on other areas of the mud flat. If care is not exercised, traffic or removal operations may mix oil with deeper sediments, exacerbating contamination.

If tides are a significant factor, evacuation operations must be coordinated with the tidal cycle. This may decrease efficiency of the operations.

Future Restoration Actions

Additional restoration may be required if sediment-dwelling organisms are severely affected by removal operations. If high levels of mortality occur and recolonization is inhibited, some type of transplantation of organisms may be considered.

2.2.6.4.3.3 Bioremediation

This restoration action is discussed in detail in Section 2.2.6.1.3.5.

2.2.7 Estuarine and Marine Subtidal Habitats

2.2.7.1 Subtidal Rock Bottoms

Subtidal rock bottom habitats include deep hard bottom environments that encompass solid, hard substrates as well as reefs composed of many individual rocks (Johnson and Pastorok, 1985). Typically in subtidal rock bottom habitats there is little or no sedimentation activity and high wave/current energy input, yielding good natural cleaning characteristics.

Restoration in subtidal rock bottom habitats in the event of oil contamination has historically consisted of minimal direct action, with primary reliance on the natural recovery process for restoration. Only one restoration action is feasible for habitat restoration:

- Natural Recovery.

The following sections summarize the available literature pertaining to restoration in subtidal rock bottom habitats.

2.2.7.1.1 Oil Related Literature

Available literature specific to the restoration of rocky subtidal habitats contaminated from an oil discharge suggests that habitat restoration can be accomplished simply by ensuring the removal of all contaminants. For many habitats this is typically not the case and additional restoration actions are generally performed to enhance the recovery process. As discussed in a report prepared for the American Petroleum Institute (Johnson and Pastorok, 1985), oil that reaches rocky bottom habitats is commonly abandoned to the natural forces of dispersal and weathering. Other alternatives for response are also presented in this report for consideration (e.g., vacuum pumping, sorption, chemical dispersal), yet the feasibility of these actions in underwater habitats is largely undetermined.

2.2.7.1.2 Non-oil Related Literature

The restoration of subtidal rock bottom habitats due to non-oil related injury is not well documented. Non-oil related injury would typically include incidents such as contamination from toxic releases other than oil as well as physical disturbances from storm activity. There is little documentation in the literature of direct restoration activities performed in rock bottom habitats related to these types of injuries.

2.2.7.1.3 Technical Feasibility of Restoration Actions

Exhibit 2.14 presents an overview of the state of technical feasibility for the restoration action appropriate to subtidal rock bottom habitats. Monitoring of natural recovery is the only technically feasible action. See Chapter 3 for a discussion of recovery.

Exhibit 2.14 Overview of technical feasibility of subtidal estuarine and marine rock bottom restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Effects**	**Legal and Administrative Factors**
Natural Recovery	Generally feasible	Generally available	Little constraint	Unlikely that additional activity be required	Coordination of monitoring activities

2.2.7.2 Subtidal Cobble-Gravel, Sand, and Silt-Mud Bottoms

With the exclusion of rock bottom areas, subtidal bottom habitats in the estuarine and marine environment have sediments that can be classified as cobble-gravel, sand, or silt-mud. Due to the similarity of restoration actions available for each of these subtidal bottom types, these alternatives are presented as one discussion, with specific attention to those applications distinct among habitats. The following restoration actions were found to be applicable to cobble-gravel, sand, and silt-mud bottom habitats:

- Natural Recovery;
- Dredging/Material Removal; and
- Sediment Containment/Replacement.

The following sections summarize the available literature on subtidal bottom restoration and discuss the technical feasibility of each restoration action.

2.2.7.2.1 Oil Related Literature

In the event of an oil discharge, bottom sediment can become contaminated by sinking of oil adhering to particulates. The available literature on oil related restoration actions for subtidal bottom habitats has focused primarily on cleanup actions. For sediment-dominated subtidal bottom habitats oil discharge cleanup actions may be used in restoration. Two studies sponsored by the American Petroleum Institute assess cleanup and restoration actions associated with oil discharge conditions. Johnson and Pastorok (1985) present an evaluation of oil discharge cleanup actions for several estuarine and marine habitat types, including subtidal bottoms. To clean oil-contaminated bottom sediments this report evaluated sediment removal. The second more recent report (API, 1991) focuses on the restoration of oil contaminated habitats and evaluates restoration alternatives applicable for subtidal habitats (i.e., sediment removal for contaminated sediments).

2.2.7.2.2 Non-oil Related Literature

Non-oil related impacts to subtidal bottom habitats typically involve the contamination of bottom sediments from toxic pollutants other than oil (e.g., PCBs, metals, etc.). The available literature on the restoration of non-oil related injury to subtidal habitats is sufficiently documented by case studies and reports that detail appropriate methods and actions to restore the contaminated bottom habitats. These reports identify sediment management practices geared toward the restoration of contaminated subtidal areas that have varying sediment characteristics.

The following literature sources are examples of studies that evaluate this information for estuarine and marine environments:

- Phillips and Malek (1987) review alternative dredging and disposal practices proposed for the restoration of contaminated sediment in Commencement Bay, Washington. Factors discussed include equipment selection and methods and the preferred dredging methods for various classes of contaminants;

- Palermo and Pankow (1988) describe appropriate dredging equipment, actions and controls for removal of contaminated sediments from an estuary. The major factors discussed include: dredging requirements, factors in selection of equipment, methodologies used to select the most appropriate equipment, operational procedures for contaminant cleanup, and control measures for resuspended sediment;

- Averett and Palermo (1989) review conceptual dredging and disposal alternatives for a contaminated estuary. The technical feasibility of alterative disposal actions such as upland and nearshore disposal is discussed, including factors such as sediment characteristics, site availability, and capacity;

- National Research Council (1989) provides a comprehensive review of the strategies surrounding the disposal of contaminated sediments, including an assessment of contamination, mobilization and resuspension, and remediation technologies;

- Palermo et al. (1989) present a strategy for the evaluation of major disposal alternatives for the disposal of contaminated sediment. This study also evaluates the dredging equipment appropriate for selected disposal alternatives, which include confined upland, confined nearshore, and contained aquatic disposal;

- Averett et al. (1990) identify feasible technologies to remove contaminated sediment from the Great Lakes. This evaluation includes a review of alternatives for the removal of contaminated sediments including subsequent transport, treatment, containment, or disposal, and those for non-removal alternatives, such as *in situ* treatment or containment of the contaminated sediment;

- Cullinane et al. (1990) provide a thorough review of alternative technologies and strategies for the removal, control, treatment, and/or disposal of contaminated dredged material. This review includes applications to and site scenarios for ocean, estuarine, and inland disposal; and

- Marcus (1991) reviews practices employed for the management of contaminated sediments in aquatic environments. The author explores current sediment regulation and alternatives for remediation.

Each of these studies focuses on the available actions for removal of contaminated sediment as well as subsequent disposal and/or treatment of the dredged material. It is likely that these practices will continue to be improved due to increasing concerns regarding the presence of contaminants in estuarine and marine environments.

2.2.7.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.15 presents an overview of the state of technical feasibility for restoration appropriate to subtidal cobble-gravel, sand, and silt-mud bottom habitats. A brief discussion of these restoration actions is presented below.

2.2.7.2.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.7.2.3.2 Dredging/Material Removal

Direct restoration of contaminated subtidal benthic environments (e.g., cobble-gravel, sand, and silt-mud) prevents continued exposure of biota to contaminants in the sediments. The USACOE categorizes sediments according to material type that includes mud, peat and organic muck, clay, silt, sand, gravel and shell, and shale (rock). The largest volume of materials dredged in the United States are sand and silt sediments, with sand, gravel, and shell sediments second in magnitude (Pequegnat et al., 1978). Organic mucks and peat sediments, while only dredged in small volumes, are found in areas with potentially more severe contamination problems (e.g., harbors and estuaries).

As identified in section 2.2.6.6.2, there are several documented cases where restoration performed for contaminated subtidal habitats was direct material removal. This activity is typically performed using one or more types of dredging equipment to remove the contaminated sediment. Sediment dredging is a well-known practice and many millions of cubic yards of sediments are dredged each year using either mechanical or hydraulic dredge equipment to maintain navigable waterways. Where material removal is performed, corresponding disposal and/or treatment actions must also be conducted to ensure proper containment and/or remediation of sediment contaminants.

Exhibit 2.15 Overview of technical feasibility of subtidal estuarine and marine cobble-gravel, sand, and silt-mud bottom restoration.

	State of Feasibility	**Availability of Services and Materials**	**Key Constraints**	**Future Restoration Actions**	**Legal and Administrative Factors**
Natural Recovery	Generally feasible	Generally available	Little constraint	Unlikely that additional activity be required	Coordination of monitoring activities
Dredging/ Material Removal	Demonstrated technically feasible	Dredging operations conducted by either federal agencies or private contractors; equipment available in most geographic regions	Effectiveness depends on material characteristics and type of dredge equipment selected; appropriate treatment and/or disposal action as must be considered	Continued presence of contamination in sediments may require further dredging activity	Dredging activities require permit from authorized agency; depending on method of disposal selected, additional permits and administrative requirements may be applicable
Sediment Capping/Replace-ment	Demonstrated as technically feasible	Capping materials generally available in most regions; equipment and transport needs met by dredging contractors and/or USACE	Improper placement of cap hinders effectiveness; short-term effects on benthic biota; long-term monitoring required	Additional sediment placement if initial cap is eroded or displaced; long-term monitoring activities required to observe containment and associated effects	Permits may be required to perform in place containment activities; coordination with oversight agencies

Availability of Services, Equipment, and Materials

The majority of dredging operations conducted in U.S. waters are performed by the USACOE, the federal agency responsible for maintaining navigable waters. The USACOE is considered the expert agency on dredging activities and typically manages and conducts dredging projects in publicly-managed waters. The USACOE maintains its own dredging fleet, comprised of several types of equipment located in the geographic areas where the USACOE manages its activities (i.e., USACOE Districts). In addition, the availability of private contractors who provide dredging services to maintain privately operated ports and harbors is equally widespread throughout the U.S.

The following describes the types of dredging equipment available to conduct sediment removal activities and characteristics of their operation.

Mechanical dredge equipment. Mechanical dredges remove bottom sediments by directly applying mechanical force to dislodge and excavate the material. Types of mechanical dredges include the clamshell, dipper, dragline, and ladder dredges.

Clamshell dredges are often used for mechanically removing sediments. This type of dredge employs a crane mounted scoop, or shovel, that has two or three "jaws" that open as the clamshell is dropped to the bottom and close together as the device is lifted. The resuspension of sediments from clamshells is relatively low, especially with better designed models that are made to be relatively water-tight after closing. Clamshell dredges typically require a barge (unpowered) or scow (powered) for transport of dredged materials.

Dipper dredges are open top shovel dredges typically used to create new harbor or channel areas by removing rocky or heavily consolidated materials rather than dredging sediments. The heavy resuspension of sediments from this type of operation makes it unacceptable for dredging contaminated sediments unless effective resuspension controls (e.g., silt curtains) are in place.

Draglines and ladder dredges are often used for mining rather than for sediment removal. Both have high levels of sediment resuspension and are not as appropriate as other types of dredges for removal of contaminated sediments (Cullinane et al., 1990).

Hydraulic dredge equipment. Hydraulic dredges remove sediment in liquid slurry form using a vacuum pump and a dredge arm or pipe extended to the bottom to vacuum material. Hydraulic dredges that do not use any specialized attachments at the sediment end of the dredge arm are known, simply, as suction dredges. The dredge arm or pipe may use a mechanism on the bottom to dislodge materials that are then suctioned through a pipeline, cutterheads and dustpans are two such attachments. Dustpans, designed for the lower Mississippi to dredge large volumes in shallow water, have a high level of resuspension and are generally inappropriate for dredging contaminated sediments (Cullinane et al., 1990). Suction dredges, with or without the cutterhead attachment, are favored hydraulic dredges for removing contaminated sediments.

Hydraulic dredges also vary in their management of sediments. Most dredges remove the sediments and either use a pipeline to transport sediments or place the materials on a barge (i.e., unpowered) or scow (i.e., self-powered) moored alongside that transports the sediment to disposal or to a shoreside settlement pond. Hopper dredges, on the other hand, are hydraulic dredges equipped with settling bins on board to allow sediments to settle out of the slurry. These bins, or hoppers, are typically filled well past overflowing, allowing the waters from which most sediments have settled to run overboard. Assuming that these waters are contaminated by their contact with the sediments, this type of overflow operation is generally not appropriate for removal of contaminated sediments.

An advantage of hopper dredges, however, is that they can be used in specialized cases (e.g., in areas of strong surface currents) where the use of anchored suction/cutterhead dredges and/or barges or pipelines may be infeasible. In cases where open water disposal of dredged material is appropriate, the hopper dredge has an advantage in situations where a down pipe (e.g., the dredge arm modified) may be needed to properly place contaminated sediments on or near the bottom using the pipeline. The hopper dredge itself (i.e., via barge) transports sediment to the disposal site with its suction pipe on board.

Constraints

Many factors must be evaluated before dredging operations can be conducted. These factors include an evaluation of appropriate dredging equipment and the subsequent disposal and/or treatment alternatives for the contaminated dredged material. In addition, dredging activity may adversely affect the contaminated habitat by destroying biota in the sediments. Thus, the environmental effects of conducting sediment removal actions must also be taken into consideration. The following paragraphs discuss each of these factors and the constraints that may be incurred during application of sediment removal actions.

When selecting the appropriate dredge equipment used in specific subtidal habitats where bottom sediments may vary in their physical characteristics, each type of equipment should be considered with respect to the level of contamination present in the sediment. For example, the primary advantage of mechanical dredges is that little additional associated water is removed with the sediments. In contrast, the contact waters that are a by-product of hydraulic dredging may represent a major disadvantage when dredging hydraulically. One major advantage of hydraulic dredging is that the resuspension of contaminated materials can be kept to a minimum. The resuspension of contaminated sediments is a primary disadvantage of mechanical dredging.

Several other factors must be considered when selecting dredge types. These include:

- Physical characteristics of the material to be dredged. Materials with a large rock composition, such as cobble-gravel sediments, are most often dredged mechanically, while organic muck (e.g., in the form of silt-mud) may not be easily removed with a clamshell but require a suction dredge;

- Quantity of material dredged. For example, clamshell operations may be slower than pipeline dredging and inappropriate for very large jobs. Hopper dredges that self-contain the sediments may be more appropriate than constructing pipelines for a small operation;

- Dredging depth and surface water characteristics. Hydraulic dredging is typically limited to 50-60 feet deep waters, clamshells may be used to 150 feet or more; clamshells, cutterhead, and suction dredges typically need calm waters in which to be anchored to work; hopper dredges can run in rough water or high currents;

- Method of disposal. Pipeline dredges may be more appropriate near shore if upland or near-shore contained disposal facilities are to be used. Hopper dredges or hopper barges may be required for long distance transport to open water disposal sites, although floating pipelines have also been used when appropriate; and

- Type of dredges available. For example, some dredge types are more prevalent in the coastal areas where currents are more of a problem and distance to open water disposal sites is farther. In more remote areas where mobilization is more difficult, the dredge types available may not be appropriate for the site conditions, thus requiring additional effort to transport the appropriate equipment.

Once excavated, the dredged material must be disposed of or used. Due to the contamination, the sediments are assumed to be below quality for any useful application (e.g., beach nourishment, fill material, road construction). It is therefore necessary to consider disposal actions for the contaminated dredged material. Current disposal methods used for such applications include upland, near-shore, or open-water disposal areas designed to effectively manage the contaminants. These disposal actions are briefly described below:

Upland Disposal. Contaminated sediments may be disposed of on shore in a designated facility. Onshore facilities must, therefore, be concerned about the potential of leaching contaminants out of the disposed sediments and into the ground water. To avoid this result, specially lined and capped facilities may be required to contain contaminants. The sediments can be transported to the lined upland unit either via a pipeline if hydraulic dredging is used, or by truck where mechanical dredging is employed. Due to the concern with contamination by associated waters in an upland unit, slurries may not be acceptable or the contaminated waters may require some treatment before discharge. If associated slurry waters are of low enough concentration they may be overflowed back into waters surrounding a dredge (i.e., via a hopper dredge operating at overflow). The contaminated sediments could then be removed from the hopper and placed in trucks for transport to an upland unit.

Near-shore, Confined Disposal. A second disposal option often employed by the USACOE in maintenance dredging operations is to construct confined disposal areas near shore. Dredged materials are placed in these sites to allow settling of the sediments and return of the associated waters to open waters. The major advantage of this option is that the units can be placed close to near-shore dredging operations and hydraulically dredged materials can be piped directly to the units for settling. A major disadvantage is that control of the associated waters is difficult with these units. If associated waters are controlled or if discharge to local near-shore waters of this contaminated water is a concern, this option may not be available. A second disadvantage is that leachate cannot be easily controlled from these units. Because the units are built with dikes in open water, after filling with sediments the lower layers of the impounded sediments will remain saturated while the upper levels will remain unsaturated. The upper levels will require capping to control leaching from precipitation. The sediments in the intermediate levels, however, will be subject to saturation and draining as tides rise and fall, and leaching contaminants from these sediments cannot be controlled, presenting long-term concerns.

Open-water Disposal. A third disposal option employed by the USACOE in the disposal of dredged sediments from maintenance dredging is to transport the materials to off-shore open-water sites designated for ocean dumping. Sediments may be transported by pipeline under certain conditions (e.g., floating pipelines may be limited in navigation lanes or very rough waters) or by hopper dredge, barge, or scow. While a primary advantage of open-water disposal is cost, another advantage is the avoidance of leachate contamination of ground water from upland units or contamination of local near-shore waters from contained disposal facilities and the associated impacts on swimming, spawning, and fishing areas and near-shore wildlife.

Obviously, there is a concern with disposing contaminated sediments and associated waters in the open ocean. Controls are available for ensuring that impacts from deep sea disposal of contaminated sediments offshore are reduced. Control of the discharge plume may be achieved using the transport pipeline, the dredge suction arm on hopper dredges (which may require modification) or some other form of down pipe to allow the correct placement of sediments and associated waters on the bottom while isolating the material in the water column during descent. This reduces entrainment and negates the effects of currents and temperature stratifications. A diffuser may be attached to the discharge end of the down pipe system to slow the release velocity and redirect the plume to release the discharge parallel to the bottom. These activities reduce resuspension of the sediments and promote mounding.

Another control used to minimize impacts from deep-water disposal of sediments is capping the mound of contaminated sediments. There is considerable research on capping that is applicable to controls for contaminated sediments (e.g., Pequegnat et al., 1978; Cullinane et al., 1990). Capping materials may consist of inert materials, chemically active material, or sealing materials. Inert materials used to cover the contaminated sediments would include either clean dredged material (i.e., dredged for this purpose or taken from maintenance dredging operations) or excavated upland materials. Capping methods and sediment replacement activities are discussed in more detail under "Sediment Containment / Replacement" (see Section 2.2.6.6.3.3).

Depending on the type of disposal selected and the severity of the contamination in the sediments, treatment may or may not be required or desired prior to disposal. If contamination levels are assumed of low severity, disposal without treatment may be sufficient, especially if disposal controls (e.g., capping) are in place. Treatment of contaminated dredged material prior to disposal is not a widespread practice and there still exist some technical constraints with various treatment alternatives to preclude them from widespread application. The primary constraint is cost, due to the fact that many treatment methods being evaluated for contaminated sediment remain in the experimental stage. Some treatment methods available for consideration in the treatment of contaminated dredged material include the following:

- Physical separation. Physical separation of contaminants from the sediments presumes that most contaminants are bound to the finer materials found in sediments (e.g., sediments with silt-mud properties). Classification of the dredged materials into coarse and fine fractions should result in a relatively concentrated fine material fraction that could be managed while the remainder of the coarse fraction is released. The cost of such operations has not been evaluated at a field level, but is expected to be substantial. However, should upland or near-shore disposal be the preferred option, the high cost of physical separation may be cost-effective in that management cost would be lowered by handling less contaminated material (Cullinane et al., 1990); and

- Contaminant extraction. This process separates contaminants from the sediments using a solvent extraction operation. This treatment application to dredged material may have potential but the current level of knowledge remains limited (Cullinane et al., 1990).

The actions for treatment and/or disposal of contaminated dredged material must be thoroughly evaluated with respect to the level of contamination and the risk of further contaminant exposure as a result of actions taken. Major issues that may be considered in contaminated sediment restoration include the following, injury associated with environmental side effects from sediment removal or treatment, selection of appropriate restoration actions in the absence of clear criteria and experimental evidence, allocation of restoration costs, and attainment of restoration goals.

<u>Future Restoration Actions</u>

Additional restoration actions may be required if dredging activities do not remove the contaminated material adequately enough to foster concurrent natural recovery processes. If sediment removal causes additional injury to the benthic community, sediment replacement may be necessary. Additional dredging may be required if the contaminated sediment was not effectively removed during the initial dredging activities. Where natural processes do not effectively dilute or bury contaminated sediment, further sediment removal operations may be needed.

Replacing sediments in subtidal benthic habitats would require that clean fill be transported to the site and placed by pipeline or surface discharge to cover the excavated area to replace the amount of sediment removed. While technically feasible, this alternative is unlikely to be necessary except in shallow waters where the change in depth would be ecologically significant. Usually in deeper waters sufficient sediment exists beneath the removed sediments to allow the natural restoration of the ecosystem. In the rare cases where the removal of sediments would expose a substrate inadequate for recolonization, some backfilling activity may be required. Natural sedimentation by wave and current activity, however, will occur in most near-shore subtidal areas and circumvent the need for backfilling.

2.2.7.2.3.3 Sediment Capping/Replacement

An alternative action to sediment removal for the restoration of contaminated sediment involves the application of in place or *in situ* controls. Possible *in situ* controls consist of containment, treatment, or combinations of the two. In practice, however, *in situ* treatment of aquatic contaminated sediments is only in the experimental stage or performed on small scales. It is not considered a viable action by most management agencies (Marcus, 1991). Sediment containment or confinement therefore is the primary focus for application of *in situ* controls.

Contaminated sediment can be contained by placing a cap over the sediments or by combining capping with lateral confining structures, such as dikes (e.g., contained aquatic disposal sites). The material used for capping typically includes clean sands or silts, which are placed on top of the contaminated sediments. Confining structures are used in cases where cap materials may be displaced, such as on a sloping surface, or disturbed by natural or man-induced activity (e.g., wave action, navigational maintenance). Contained aquatic sites constructed to confine contaminated sediments also help ensure that capping materials are properly placed and they effectively cover the contaminated sediments.

Sediment confinement is only considered an appropriate restoration alternative under certain circumstances. These include:

- If natural recovery, or no-action, does not provide effective dispersement of contaminants;
- If the source of pollutant discharge is contained;
- If constraints of conducting sediment removal activities are too great (e.g., cost, environmental effects);
- If sufficient capping material is available; and
- If the site will not be unreasonably disturbed by natural or human intrusion (e.g., hydrological factors, dredging) (Marcus, 1991).

Availability of Services, Equipment, and Materials

Capping materials may include clean dredged sediments from a maintenance dredging operation or material that is excavated from an upland site. Typical capping operations include the placement of suitable materials over the sediments using a ratio of clean material to contaminated sediment. Based on communication with and published sources from the USACOE, a generally accepted ratio of capping material to contaminated sediment for an adequate cap on contaminated sediment ranges from three to five parts clean material to one part contaminated (USACOE, 1989; Averett and Palermo, 1989; Holliday, 1992). Clean dredged material is a preferred capping material due to its similar composition to contaminated bottom sediment (in any given area), as well as the ease of acquisition, transport, and placement of such materials. Capping operations may be planned to coordinate with maintenance dredging operations so that clean dredged material may be used in the cap.

The availability of clean dredged material and other sources of suitable fill material will vary by geographic location. The availability of clean material for use in capping operations often depends upon the schedule for maintenance dredging whereby fill material is produced or may rely on access to upland sites for material. Also, clean dredged material is often used in beneficial use applications (e.g., beach nourishment) and, therefore, significant quantities of available material may be earmarked for this type of operation. In this case, additional costs may be incurred to obtain material from sources other than maintenance dredging operations. It is common practice for capping operations to use clean material which is located nearby the contaminated site in order to defray costs of transport.

If clean material is obtained from maintenance dredging operations, capping activities may be scheduled to coordinate with the maintenance dredging operations so that additional equipment requirements are not needed to move and place the capping material. If suitable capping material is provided from upland sources, equipment requirements needed to perform site containment include material transport from source to the contaminated aquatic site. These activities would typically involve mobile transport of the material to a barge equipped with necessary controls for placement of material onto the contaminated bottom sediment. The availability of such equipment is widespread in most regions with marine and estuarine resources and may be contracted either from federal agencies such as the USACOE or from private contractors who specialize in dredging operations.

Constraints

One advantage of capping contaminated sediments as a restoration action is that materials are not resuspended into the aquatic environment as they can be when sediments are removed. Also, surrounding benthic organisms are prevented or restricted from contact with the contaminated sediments after placement of the capping material. A disadvantage of *in situ* capping, however, is that a large surface area of bottom sediments may require capping, thereby requiring the placement of large quantities of clean material. The placement of such large quantities of material on the local benthic environment may cause some environmental detriment in the short-term. Another logistical disadvantage, and one very important to the selection of the preferred restoration alternative, is that *in situ* capping cannot be used in an area where the cap may be disturbed either by natural forces (e.g., major storms or earthquakes and slides) or anthropomorphic activities (e.g., shipping, maintenance dredging, mining) (Averett and Palermo, 1989; Averett et al., 1990).

Additional constraints related to capping operations include problems associated with the inaccurate emplacement of materials on the habitat bottom and the potential for erosion processes to alter the effectiveness of the cap. Specialized equipment is available to minimize problems associated with misdirected capping material so that the initial cap is effectively placed. Also, it is essential to develop long-term monitoring procedures to detect erosion and ensure that the contaminants do not bioaccumulate in the biota (Averett and Palermo, 1989; Marcus, 1991).

Future Restoration Actions

As identified above, long-term monitoring should be conducted to observe the effectiveness of the cap and determine additional management procedures based on results of initial site containment.

2.2.8 Riverine and Lacustrine Shorelines

The following discusses restoration actions and the related technical feasibility of restoration for riverine and lacustrine (lake) freshwater habitats.

2.2.8.1 Rocky Shores

- Natural Recovery;
- Sandblasting;
- Steam Cleaning;
- Flushing; and
- Bioremediation.

2.2.8.1.1 Oil Related Literature

The same literature sources used in the evaluation of the technical feasibility of restoration actions in intertidal rocky shore habitats (Section 2.2.6.1) are applicable to riverine and lacustrine rocky shore habitats. In addition to the sources detailed in the intertidal section, Foley and Tresidder (1977) evaluated pressure washing and steam cleaning in freshwater rock shore environments. Fremling (1981) also evaluated pressure washing of rip rap shorelines in a lacustrine environment.

2.2.8.1.2 Non-oil Related Literature

The literature that discusses restoration in rocky shore intertidal habitats (see Section 2.2.6.1) was also used to evaluate technical feasibility in riverine and lacustrine rock shorelines. This literature deals primarily with oil contamination.

2.2.8.1.3 Technical Feasibility of Restoration Actions

The feasibility of each action is summarized in the previously-presented Exhibit 2.10 and is discussed below. In general, restoration of riverine and lacustrine shorelines is subject to the same feasibility issues as similar estuarine and marine intertidal habitats.

2.2.8.1.3.1 Natural Recovery

Monitoring of natural recovery is always technically feasible. See Chapter 3 for discussion of recovery.

2.2.8.1.3.2 Sandblasting

The technical feasibility of sandblasting in riverine and lacustrine rocky shore habitats is similar to that in rocky intertidal habitats. See Section 2.2.6.1.3.2. for a detailed discussion.

2.2.8.1.3.3 Steam Cleaning

In addition to the discussion provided in Section 2.2.6.1.3.2. regarding the literature concerning steam cleaning in rocky marine intertidal habitats, Foley and Tresidder (1977) report the use of steam cleaning of rock, steel, and wood surfaces following the *Nepco 140* barge oil discharge in the St. Lawrence river in 1976. Steam cleaning was used to remove residual oil stains on rock shores and manmade structures after initial cleaning was conducted. While this activity was used as a restoration action, it was used in conjunction with open water and shoreline cleanup recovery efforts.

The technical feasibility of steam cleaning in riverine and lacustrine rocky shore habitats is similar to that rocky intertidal habitats. See Section 2.2.6.1.3.3. for a detailed discussion.

2.2.8.1.3.4 Flushing

In addition to the literature discussed previously for flushing in estuarine and marine intertidal habitats, Foley and Tresidder (1977) and Fremling (1981) discuss the use of pressure washing in riverine and lacustrine environments. The activities described in both these sources are high pressure spraying. Foley and Tresidder describe the use of "water blasting" on rock, steel, and wood surfaces following cleanup activities subsequent to the *Nepco 140* barge discharge in the St. Lawrence River. Fremling notes that high pressure sprays were used in an attempt to remove the 18-inch-wide "tar-like fraction" from rip rap sections along the 3.6-mile perimeter of Lake Winona following the long-term release of heating oil.

The technical feasibility of flushing in riverine and lacustrine rocky shore habitats is similar to that in rocky intertidal habitats. See Section 2.2.6.1.3.3. for a detailed discussion.

2.2.8.1.3.5 Bioremediation

The technical feasibility of bioremediation in riverine and lacustrine rocky shore habitats is similar to that in rocky intertidal habitats. See Section 2.2.6.1.3.3. for a detailed discussion.

2.2.8.2 Cobble-Gravel Shores

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation; and
- Bioremediation.

2.2.8.2.1 Oil Related Literature

The same literature sources used in the evaluation of the technical feasibility of restoration actions in intertidal cobble-gravel shore habitats (Section 2.2.6.1) are used for similar riverine and lacustrine habitats. In addition, the observations of Little and Little (1991) regarding the restoration efforts of rock and cobble shores were evaluated.

2.2.8.2.2 Non-oil Related Literature

The literature related to the restoration of cobble and gravel shorelines is primarily oil discharge related.

2.2.8.2.3 Technical Feasibility of Restoration Actions

The feasibility of each restoration action is similar to that for estuarine and marine cobble-gravel shores. Technical feasibility of actions were previously summaried in Exhibit 2.11 and Section 2.2.6.2.3.

2.2.8.3 Sand Shores

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation;
- Bioremediation; and
- Incineration.

2.2.8.3.1 Oil Related Literature

The oil discharge related literature used to examine technical feasibility of restoration actions in freshwater sand shoreline environments is the same as that used to evaluate feasibility in intertidal sand shore habitats (see Section 2.2.6.). In addition to the sources detailed in that section, Fremling (1983) provided details on pressure washing of oil stains in a lacustrine environment.

2.2.8.3.2 Non-oil Related Literature

The literature that deals with restoration of sand shore habitats is primarily oil discharge related.

2.2.8.3.3 Technical Feasibility of Restoration Actions

The technical feasibility of each action is similar to that for estuarine and marine sand shores, as summarized in Exhibit 2.12. See Section 2.2.6.3.3 for discussion.

2.2.8.4 Silt-Mud Shore

- Natural Recovery;
- Sediment Removal/Replacement; and
- Bioremediation.

2.2.8.4.1 Oil Related Literature

The oil discharge related literature used to evaluate the technical feasibility of silt-mud shoreline restoration in riverine and lacustrine environments is the same as that used for mud flat intertidal environments (see Section 2.2.6.4.1.). In addition to these sources, Smith (1987) and the American Petroleum Institute (1991) were used as references for sediment removal and replacement operations.

2.2.8.4.2 Non-oil Related Literature

The literature dealing with the restoration of silt-mud shorelines is primarily oil discharge related.

2.2.8.4.3 Technical Feasibility of Restoration Actions

The technical feasibility of restoration actions is similar to that for estuarine and marine mud flats, as summarized in Exhibit 2.13. See Section 2.2.6.4.3 for discussion in addition to that below.

2.2.8.4.3.1 Natural Recovery

Monitoring of natural recovery is always technically feasible. See Chapter 3 for a discussion of recovery.

2.2.8.4.3.2 Sediment Removal/Replacement

Smith (1987) documents the use of sediment removal and replacement as a restoration action for a silt-mud shoreline of a lake in Portland, Oregon. An oil discharge occurred in 1985, which was caused by a separator pond malfunction at a waste oil treatment and recycling facility. In the course of the cleanup, the water level dropped one foot, leaving stranded oil in a band on the shoreline approximately 10 feet wide.

The restoration effort consisted of topsoil removal and replacement on the shoreline, along with planting of grass. Topsoil had been removed to a depth of 2 inches over an area from the shoreline to a point where no further oil was discernible. Topsoil was replaced using material from a local supplier. After the topsoil was spread, it was seeded with fescue grass. Sludge from a paper mill was suggested as a soil amendment to replace lost humus-rich soil. However, concern was raised about the use of the sludge and, at it was decided to replace the oil-contaminated soil with common topsoil.

The American Petroleum Institute (1991) developed a restoration scenario for conditions following a discharge of gasoline into the high energy Wolf Lodge Creek. Restoration conducted following the discharge consisted of streambed agitation and is discussed in Section 2.2.9.2.3.3. API suggests a restoration scenario that includes the manual and mechanical removal and replacement of streambank soils in addition to streambed agitation.

The technical feasibility for the removal and replacement of contaminated silt-mud shores is the same as for intertidal mud flat habitats. Refer to Section 2.2.6.4.3.3. for a further explanation of the factors affecting removal and replacement.

2.2.8.4.3.3 Bioremediation

Bioremediation of freshwater silt-mud shorelines is similar to that in intertidal mud flat habitats. The discussion of technical feasibility found in Section 2.2.6.4.3.3 applies to riverine and lacustrine environments as well.

2.2.9 Riverine Bottom

The following section summarizes restoration actions for riverine bottom environments.

2.2.9.1 Rock Bottoms

As identified in Section 2.2.7.1. for estuarine and marine subtidal rock bottom habitats, the restoration action applicable to these habitats is natural recovery. Due to limited available literature on the restoration of rock bottom river and stream habitats injured by pollutants and the similarity of this habitat to estuarine and marine rock bottom habitats, monitoring of natural recovery is the only feasible action for river and stream rock bottom habitats. Refer to Section 2.2.7.1. for a discussion of the technical feasibility of this restoration action.

2.2.9.2 Cobble-Gravel, Sand, and Silt-Mud Bottoms

Restoration actions for riverine cobble-gravel, sand, and silt-mud bottom habitats injured by contaminants are similar to two actions described above for subtidal estuarine and marine habitats (see Section 2.2.7.2.). One additional action, sediment agitation, is also considered for riverine habitats. For these habitats, restoration actions include the following:

- Natural Recovery;
- Dredging/Sediment Removal; and
- Sediment Agitation.

The following sections summarize available literature related to river and stream restoration and discuss the technical feasibility of each restoration action.

2.2.9.2.1 Oil Related Literature

Cases of oil discharge related restoration of river and stream bed habitats are not as well documented in the literature as those involving sediment contamination in estuaries or marine habitats. One case involving a fuel discharge in the Savannah River (Brown, 1989) identified the effects of oiling to be minimal in bottom sediments since the remaining surface oil after cleanup was left to natural dispersion. Adverse impacts from this discharge focused primarily on injuries to wetlands, waterfowl, shellfish, and other vegetation. Two other restoration cases studies identified in the literature refer to a gasoline discharge located in a Northern Idaho creek (Graves, 1985; API, 1991). Creek restoration was performed using a stream agitation action, a method that is technically feasible in shallow water habitats.

2.2.9.2.2 Non-oil Related Literature

Non-oil related impacts can include stresses on the habitat due to the deterioration of water quality (i.e., from temperature changes, excessive turbidity), substrate modification, flow fluctuations, and biotic interactions. The restoration of rivers and streams affected by non-oil related impacts is documented in the following literature sources:

- Bechly (1981) describes a case study of the restoration efforts performed in the Cowlitz and Columbia Rivers after the Mount St. Helens volcanic eruption. Excavation of large amounts of sediment was performed in both rivers;

- Institute of Environmental Sciences (1982) evaluates the George Palmiter method of river restoration. This method was designed as a labor intensive method of preventing erosion and flooding;

- Herricks and Osborne (1985) discuss the restoration and protection of water quality in streams and rivers. This chapter identifies the uses and impacts of restoration and discusses general approaches to restoration and protection;

- Starnes (1985) presents an overview of stream reclamation approaches and case studies where coal mining related impacts were restored. These approaches include methods of instream habitat restoration;

- Gore et al. (1988) summarize methods of river and stream restoration and identify the need to eliminate pollutant load in surface runoff, control erosion, and sustain faunal habitats;

- National Research Council (NRC, 1992) presents a thorough assessment of river and stream restoration, identifies case studies of historical restoration projects, and evaluates habitat functions, stresses, and effective management actions.

Restoration actions identified for non-oil related impact to river and stream habitats emphasize actions for the rehabilitation of ecosystem impacts related to increased sediment loads, poor water quality, and declines of habitat species. These injuries can be restored through actions which allow dilution or transfer, removal, or isolation of the pollutants.

2.2.9.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.16 presents an overview of the technical feasibility of the restoration actions appropriate to riverine cobble-gravel, sand, and silt-mud bottom habitats. A brief discussion of these restoration actions are presented below.

2.2.9.2.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.9.2.3.2 Dredging/Sediment Removal

Sediment removal using dredging actions to eliminate contaminated bottom sediment is a technically feasible approach for subtidal river and stream habitats (Herricks and Osborne, 1985; NRC, 1992). River dredging is a common method used to maintain navigational waterways. However, this practice is not as common for use in smaller streams. The technical feasibility of dredging and replacement of bottom sediments as a restoration action was discussed above for estuarine and marine subtidal habitats (see Section 2.2.7.2.3.2). These factors are also applicable to riverine bottoms.

2.2.9.2.3.3 Sediment Agitation

A restoration action applicable to shallow river and stream habitats is stream bed agitation. Graves (1985) describes the application of stream bed agitation after a gasoline discharge in Wolf Lodge Creek, Idaho. This restoration action is also identified in API (1991). In this application, officials concluded that after the initial cleanup, additional restoration was necessary because some of the dischargeed gasoline had been trapped in the stream bed underneath gravel and debris. Gasoline continued to leach from these areas contaminating the creek waters.

Exhibit 2.16 Overview of technical feasibility of riverine cobble-gravel, sand, and silt-mud bottom restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Actions	Legal and Administrative Factors
Natural Recovery	Generally feasible	Generally available	Little constraint	Unlikely that additional activity be required	Coordination of monitoring activities
Dredging/Sediment Removal	Demonstrated technically feasible	Dredging operations conducted by either federal agencies or private contractors; equipment available in most geographic regions	Effectiveness depends on material characteristics and type of dredge equipment selected; appropriate treatment and/or disposal action as must be considered	Continued presence of contamination in sediments may require further dredging activity	Dredging activities require permit from authorized agency; depending on method of disposal selected, additional permits and administrative requirements may be applicable
Sediment Agitation	Demonstrated as technically feasible	Heavy equipment, labor, and materials generally available	Feasible only in shallow water areas	Potential need for vegetation and additional soil removal if contamination poses long-term threat	Little constraint; coordination of activities with appropriate authorities

Stream bed agitation was applied in an attempt to release gasoline trapped in the stream bed. A bulldozer was used to agitate the gravel creek bed by dragging the blade backward throughout the entire stream bed. A tightly wound chain link fence was attached to the bottom of the bulldozer blade to smooth the stirred stream bed and to facilitate agitation of small gravel and debris. Three inches of stream bed were turned over by dragging the bulldozer blade. Sorbent blankets were deployed at about one-quarter mile intervals in slow-moving areas of the stream to capture released gasoline. Sorbent and contaminant boom were placed downstream from the agitation area to capture any gasoline that was not removed by the sorbent blankets.

Availability of Services, Equipment, and Materials

Equipment requirements for the streambed agitation action include the use of a bulldozer with rake attachments and sorbent materials to contain remaining pollutants once agitation is implemented. Heavy machinery and trained operators are typically available through local private contractors. Materials used to absorb excess pollutants can generally be obtained through local discharge response agencies or contractors.

Constraints

The case study identified above concluded that stream bed agitation appears to be a technically feasible method of removing gasoline trapped in shallow stream bed sediments (Graves, 1985). The action, however, is only applicable to shallow streams with low to moderate current that allow the bulldozer to operate. The stream bed in which it was applied consisted of gravel. It was not attempted in an area of the stream where the bottom was silt because there was concern that stirring up too much sediment would have an adverse effect on the stream.

Future Restoration Actions

Additional restoration actions following the streambed agitation action that may be feasible and warranted include removal of injured riparian vegetation and contaminated streambank soils. These activities would be necessary in cases where there is potential for significant long-term impacts from the pollutant.

2.2.10 Lacustrine Bottom

The following summarizes restoration actions for lacustrine bottom environments.

2.2.10.1 Rock Bottom

As identified in Section 2.2.7.1. for estuarine and marine subtidal, and Section 2.2.9.1 for riverine, rock bottom habitats, the only restoration action applicable to these habitats is monitoring of natural recovery. Lacustrine rock bottoms are assumed to be similar.

2.2.10.2 Cobble-Gravel, Sand, and Silt-Mud Bottom

Restoration actions for lacustrine cobble-gravel, sand, and silt-mud bottom habitats injured by contaminants are similar to those actions described above for subtidal estuarine and marine habitats (see Section 2.2.7.2.). For these habitats, restoration actions include:

- Natural Recovery;
- Dredging/Sediment Removal; and
- Sediment Capping.

The following sections summarize available literature related to lake restoration and discuss the technical feasibility of each restoration action.

2.2.10.2.1 Oil Related Literature

Oil related lake restoration is not extensively documented in the literature. One case study identified oil discharge cleanup actions performed in Lake Winona, Minnesota, due to a fuel discharge (Fremling, 1981). Post-cleanup actions involved artificial circulation of the lake to purge the lake of residual oil. Contacts with scientific experts regarding ongoing lake restoration actions also confirmed that the science of oil-related restoration is not widely developed or documented (Peterson, 1993; Lazorchak, 1993).

2.2.10.2.2 Non-oil Related Literature

Restoration actions for lakes degraded by non-oil related factors are typically employed to modify lake water quality and shift the lake system closer to its original state. Non-oil related impacts to lacustrine systems that may warrant restoration actions include the presence of high levels of nutrients in the sediment, excessive sedimentation, the presence of toxic materials other than oil in the sediment, and increased aquatic macrophyte growth.

The most common restoration actions include sediment removal using dredging equipment and sediment covering (i.e., capping) to contain or control the source of degradation (e.g., presence of toxic sediment, excess nutrient releases). The following literature sources identify case studies where these actions were employed in lacustrine habitats and evaluate management strategies associated with each.

- Peterson (1979) addresses the positive and negative aspects of dredging freshwater lakes and evaluates the types of effective dredge equipment for sediment removal. Examples of successful dredging projects performed in lake systems are also presented;

- Cooke (1980) evaluates the process of covering bottom sediments as a restoration action to control macrophytes and sediment nutrient release;

- Welch (1981) describes the dilution/flushing action used in eutrophic lacustrine systems to alter the high nutrient content. Case studies of this action are presented;

- Peterson (1982) presents information of the effectiveness of sediment removal as a lake restoration action. This includes an evaluation of the action, considerations for sediment removal, and case histories where this action has been employed;

- Cooke (1983) reviews several lake restoration actions for use in lake systems. These include sediment removal, nutrient/silt diversion, dilution/flushing, phosphorus inactivation, and sediment covers;

- Welch and Cooke (1987) evaluate lake management actions which address the restoration of lakes with poor water quality;

- Bjork (1988) presents a summary of several lake restoration case studies which employed actions such as sediment removal and *in situ* sediment capping to control and immobilize problem elements in the system;

- Environmental Protection Agency (1988b) presents a review of effective in-lake restoration actions which have been found to be effective, long-lasting, and generally without significant negative impact when used properly. Sediment removal is evaluated in this review; and

- National Research Council (NRC, 1992) presents a chapter on lake restoration which evaluates the range of stresses imposed on lacustrine systems, and the various actions used to restore lake quality to its natural state. This review identifies sediment removal as the available method to restore lakes degraded by toxic sediments.

2.2.10.2.3 Technical Feasibility of Restoration Actions

Exhibit 2.17 presents an overview of the state of technical feasibility for the restoration actions appropriate to lacustrine cobble-gravel, sand, and silt-mud bottom habitats. A brief discussion of these actions are presented below.

2.2.10.2.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.2.10.2.3.2 Dredging/Sediment Removal

Sediment removal using dredging actions to eliminate contaminated bottom sediment is a technically feasible approach for lacustrine habitats (Peterson, 1978; Cooke, 1983; Peterson, 1982; Bjork, 1988; EPA, 1988b; NRC, 1992). Sediment removal is one of the most commonly prescribed actions for long-term lake improvement. Its main purposes are to remove toxic materials, macrophytes, and nutrient-rich sediments as well as to deepen lakes. The technical feasibility of dredging and replacement of bottom sediments as a restoration action was discussed above for estuarine and marine subtidal habitats. These factors are also applicable to lacustrine habitats (see Section 2.2.7.2.3.2).

2.2.10.2.3.3 Sediment Capping/Replacement

As discussed for subtidal estuarine and marine bottom habitats, sediment capping is a technically feasible action for the containment of contaminated sediment. This method of restoration, as a contaminant control measure, is widely practiced and evaluated and provides an effective and economical action for managing contaminated bottom sediments and for the prevention of macrophyte growth in lakes (Cooke, 1980, 1983; Bjork, 1988; Averett et al., 1990). The technical feasibility of a sediment cap is dependant upon specific-site conditions. Refer to Section 2.2.7.2.3.3 for further discussion of these factors.

Exhibit 2.17 Overview of technical feasibility of lacustrine cobble-gravel, sand, and silt-mud bottom restoration.

	State of Feasibility	Availability of Services and Materials	Key Constraints	Future Restoration Actions	Legal and Administrative Factors
Natural Recovery	Generally feasible; favorable environmental conditions improve effectiveness	Generally available	Little constraint	Unlikely that additional activity be required	Coordination of monitoring activities
Dredging/Sediment Removal	Demonstrated technically feasible; effectiveness varies based on site conditions and type of equipment used	Dredging operations conducted by either federal agencies or private contractors; equipment available in most geographic regions	Effectiveness depends on material characteristics and type of dredge equipment selected; appropriate treatment and/or disposal action a must be considered	Continued presence of contamination in sediments may require further dredging activity	Dredging activities require permit from authorized agency; depending on method of disposal selected, additional permits and administrative requirements may be applicable
Sediment Capping/ Replacement	Demonstrated as technically feasible; selection of this alternative depends on site conditions and related factors	Capping materials generally available in most regions; equipment and transport needs met by dredging contractors and/or USACE	Improper placement of cap hinders effectiveness; short-term effects on benthic biota; long-term monitoring required	Additional sediment placement if initial cap is eroded or displaced; long-term monitoring activities required to observe containment and associated effects	Permits may be required to perform in place containment activities; coordination with oversight agencies

2.3 Biological Natural Resource Restoration

In addition to habitat restoration, fish and wildlife populations that live in these habitats may also require restoration. Several technically feasible restoration alternatives exist. Restoration actions typically include natural recovery monitoring, restocking, and various types of habitat enhancement, protection, and management practices.

Natural recovery, or no action (except monitoring), is typically used when no other restoration actions exist or would cause more injury if implemented. All actions require periodic monitoring of the area to ensure that recovery is occurring as expected.

The objective of restocking is to facilitate the recovery process by introducing or stocking species the same as or comparable to those injured. Although restocking is beneficial in many situations, there are potential problems and disadvantages resulting from the process and it may not be successful. These issues are discussed in Section 3.3.

Each of the following subsections: summarizes the oil discharge and non-oil discharge related literature, briefly describes each restoration action and discusses the technical feasibility of each action for shellfish (Section 2.3.1), fish (Section 2.3.2), reptiles (Section 2.3.3), birds (Section 2.3.4), and mammals (Section 2.3.5).

2.3.1 Shellfish

The restoration actions for restoring shellfish populations include:

- Natural Recovery;
- Reef Reconstruction;
- Hatchery and Seeding of Beds (restocking);
- Habitat Restoration and Enhancement;
- Modification of Fishery Management Practices; and
- Habitat Protection and Acquisition.

Monitoring is always feasible. A complete discussion of the technical feasibility of mollusc reef reconstruction is provided in Section 2.2.4 (Mollusc Reefs).

Hatchery and seeding programs exist for other types of shellfish and invertebrates. A number of states have seeding programs for clams and other molluscs. For example, Washington state plants hatchery-raised juvenile geoduck clams throughout Puget Sound. As the actions are generally technically feasible, the choice of the seeding alternative is dependent on effectiveness and success, as well as cost, discussed in detail in Sections 3.3 and 4.3, respectively.

The other restorations for shellfish are analogous to those for fish. See below for discussion of these actions.

2.3.2 Fish

Five general approaches have been used and documented as technically feasible for restoring injured fish populations. These actions include:

- Natural Recovery;
- Restocking/Replacement;
- Habitat Restoration and Enhancement;
- Modification of Fishery Management Practices; and
- Habitat Protection and Acquisition.

Habitat restoration and enhancement consists of improving the infrastructure of the habitat used by the fish surviving contamination. There are many forms of habitat enhancement, including construction of artificial reefs, development of spawning channels, construction of stream channel modifications, initiation of liming programs for acidic river environments, and improvement of fish passageways. These actions may be mitigating measures for the injury caused by oil discharges.

Modification of fishery management practices includes the initiation of policies that temporarily reduce or eliminate recreational and commercial harvesting of specific fisheries injured by contamination. The object is to allow the fishery population to recover from the effects of contamination without negative interference from harvesting.

Habitat protection and acquisition consists of designating areas as off-limits for human uses that would otherwise be open. The objective is to facilitate recovery of injured populations.

It is important to recognize that the selection of actions may differ depending upon whether the emphasis is on restoring the fish populations or the services provided by the fishery. The focus in this document is on the former, with the assumption that services will follow. However, services might be restored by replacement alternatives, such as providing additional fisheries or fishing areas.

The technical feasibility of each restoration action is described in greater detail below. Refer to Chapter 3 for discussions of effectiveness and success.

2.3.2.1 Oil Related Literature

Following the *Exxon Valdez* oil discharge, several reports were developed that describe restoration actions proposed for natural resources injured by the discharge. One such document by the Exxon Valdez Oil Spill Trustees (1992b) provides a summary of the habitats and species injured and gives brief descriptions of several potential restoration actions for these resources. Also included in the report is the rationale behind rejecting certain restoration actions previously considered. This report does not represent the final and complete restoration plan for the *Exxon Valdez* discharge, but it does represent the most recent and comprehensive oil discharge restoration plan available. This plan provides several restoration actions for injured fisheries, including intensifying or implementing recreational and commercial fishery management practices, enhancing fishery habitats (e.g., improvement of spawning substrates and establishment of alternative salmon runs), and eliminating sources of persistent contamination of spawning substrates.

Cairns and Buikema (1984) discuss the importance, vulnerability, and recovery potential of various natural resources susceptible to adverse impacts from oil discharges. They provide insight on some issues related to the restoration of fisheries injured by an oil discharge. In addition to suggested methods of assessing the impact of a discharge on fisheries, several fishery restoration actions are recommended including natural recovery monitoring, hydroelectric dam fish ladders, removal of massive pollutant sources, and the control of habitat invaders (e.g., sea lampreys).

2.3.2.2 Non-oil Related Literature

Section 3.3.2 contains a detailed discussion of restoration actions for fish populations. It focuses on effectiveness and success of technically feasible actions. The following is an overview of feasible actions.

Most fishery restoration actions relate to general restocking and hatchery research. Since this science is relatively well developed and documented, more discussion of the findings is provided here than for other resources.

Smith et al. (1990) describe the hatchery production process of advanced juveniles (phase II) and subadults/adults (phase III) striped bass and striped bass hybrids in earthen ponds. They provide information on pond design, pre-stocking pond preparation, acclimation and estimation of mortalities, stocking densities, feeding, monitoring, growth and survival, water quality sampling management, predators and competitors, diseases, vegetation, and harvesting.

The "Lake Superior Annual Report" for 1987, compiled by the state of Minnesota, discusses changes in the fish populations of Lake Superior and provides information on the monitoring, restocking, and other control methods of these fish populations. The changes in the fisheries of Lake Superior are in part caused by excessive harvesting and the introduction of several new species, including the sea lamprey, rainbow trout, and Pacific salmon. The report focuses on the impact of the sea lamprey on lake trout and the subsequent attempts to restore the lake trout population. Methods of reestablishing the lake trout fishery through controlling the sea lamprey, limiting commercial harvest, and stocking the lake with juvenile lake trout are described in this report.

In 1978, Nelson et al., of Enviro Control, Inc., prepared a handbook sponsored by the U.S. Department of the Interior which summarizes almost 300 fish and wildlife habitat and population improvement actions. The alternatives discussed include enhancement actions proven effective during previous dam and reservoir projects or determined to be potentially effective by experts in the field. A brief summary of each action provides engineering features, hydrological effects, biological effects, relative costs, and references. The fish and wildlife habitat improvement actions are reservoir flood basins, reservoir conservation pools, dam discharge systems, streamflows, riffles, and pools, streamside protection, and general practices. The fish and wildlife population improvement actions are fish propagation, fish passage, fish stocking and control, wildlife propagation and control, and wildlife protection at canals.

Bell et al. (1989) evaluate the biological, physical, and economic effectiveness of eight manufactured artificial reef structures. These structures were tested at sites off the coast of South Carolina as part of the state's Marine Artificial Reef Program. Although the evaluation is on-going to assess long-term effects, observation within the first three years of the study led to several preliminary conclusions and recommendations. Bell et al. describe the background of South Carolina's Marine Artificial Reef Program, methodology used for this study, specifications of the eight manufactured reef structures tested, economic cost of each reef structure type, and preliminary results and conclusions of the study.

Prince and Maughan (1978) present and discuss several biological and cost issues relevant to the development of freshwater artificial reefs. The biological issues addressed include fish abundance, fish colonization, fish harvest rates, and fish production in freshwater environments in relation to the existence of artificial reefs. The discussion on cost issues emphasized the possibility of using donated equipment, supplies, and labor to construct artificial reefs. This discussion was based on an actual artificial reef development program for Smith Mountain Lake in Virginia.

Feigenbaum et al. (1989) present methodologies, results, and conclusions from a three-year artificial reef study program in the Chesapeake Bay supported by a mitigation fund. The study experimented with various reef structures and sites. The stress levels and stability of the structures were tested by placing them in both the bay and nearby coastal waters. Feigenbaum et al. (1989) also present success rates of the various reef structures and sites for attracting fish populations and increasing catch rates. Recommendations of the best structural types and reef locations were derived based on the results of the study.

Duedall and Champ (1991) provide an international viewpoint of artificial reef design and construction. They discuss the various groups currently involved in the design and construction of artificial reefs, common materials used in reef construction worldwide, various functions of artificial reefs, biological benefits derived from reefs, factors involved in selecting an appropriate artificial reef site, and new developments of artificial reefs in Japan.

Hueckel et al. (1989) describe a mitigation project in Washington that involved the construction of an artificial reef. The reef was developed in a nearby sand bottom area to mitigate the loss of an area of rocky subtidal habitat destroyed from a shoreline fill project. One-half of the sand bottom area (2.83 ha) was covered with 181,400 metric tons of quarry rock ranging in size from 0.3 meters to 1.2 meters in diameter. The reef structures were placed approximately 15 meters apart.

Knatz (1987) describes three projects under consideration as mitigation for port landfill development in Southern California. One project consists of constructing an artificial reef near the Port of Long Beach under the guidelines of state and federal wildlife agencies. The other projects under consideration are two wetland habitat enhancement projects near the port. The determination of adequate mitigation of a development project and the concept of mitigation banking are discussed. The relative technical concerns and cost estimates are provided for each project.

McGurrin and Fedler (1989) evaluate the planning, siting, and socio-economic impacts associated with the rigs-to-reefs development program, specifically the Tenneco II artificial reef project. This project consisted of transporting three obsolete petroleum platforms from Louisiana to south Florida. The platforms now serve as a large artificial reef site for recreational fishermen.

Frissell and Nawa (1992) present the results from a study conducted on fishery habitat enhancement with artificial stream structures in Oregon and Washington. Various stream structure types were placed at several project sites and evaluated to determine the rates and possible causes of deterioration of each structure. The results revealed no direct correlation between the rate of failure and structural design. However, the characteristics of the stream in which the artificial structure was located had some relationship with the rate of structural failure. Frissell and Nawa provide conclusions and recommendations about the success and effectiveness of artificial stream structures developed from results of their study.

Smallowitz (1989) discusses the effects that the increasing number of hydroelectric dams in the Northwest have had on the annual runs of salmon and trout. The program to alleviate the injury inflicted on these migrating fish populations was initiated by the Northwest Power Act. The program includes both the enforcement of management practice policies and the installation of mechanical fish passageways around or through the dams. Several demonstrated fish passageway improvement actions are described.

Gore et al. (1988) summarize the issues and alternatives associated with the restoration of rivers and streams. Some of the considerations include proper hydrology, improved water quality, adequate riparian vegetation, appropriate distribution of macroinvertebrates, and adequate planning and monitoring of the restoration effort. After most of the river or stream infrastructure is established, efforts can be concentrated on the enhancement of fish habitats. Gore, et al. (1988) suggest the use of artificial stream structures based on a literature review. These structures include various current deflectors, dams, boulder placements, trash catchers, and bank covers.

Wesche (1985) discusses many aspects of river and stream restoration often required following channel modification, including a description of the impacts on the habitat, and guidelines for the planning, application, construction, and installation of various reclamation structures (i.e., dams, deflectors) and other actions (i.e., substrate development, bank cover treatments). These river and stream-based restoration actions are also discussed in relation to the enhancement of associated fish habitats.

Liming of an acidified waterway is a habitat enhancement/restoration action that can be used to mitigate oil injuries to fish. Watt (1986) describes a small liming program established to reduce the effects of acidity on the salmon populations that inhabit several rivers in Nova Scotia. Chemical transportation on the rivers has caused the pH to decline. The restoration action presented as technically feasible in this situation is the addition of limestone to the rivers to counteract the acidic contamination. This same action can also be used on streams and lakes with low pH levels. In addition to describing the liming process, the expected benefits from the liming program are discussed.

2.3.2.3 Technical Feasibility of Restoration Actions

The following sections discuss the technical feasibility of fishery restoration actions for fish populations injured by oil discharges and associated contamination.

2.3.2.3.1 Natural Recovery

Monitoring of natural recovery is always feasible. See Chapter 3 for a discussion of recovery.

2.3.2.3.2 Restocking/Replacement

See Section 3.3 for a description of various research on this action.

Availability of Services, Materials, and Equipment

Only certain species of fish are readily available for restocking purposes from private, tribal, and public hatcheries. These hatcheries usually concentrate on growing important game fish species (i.e., trout, salmon). However, less popular, non-game species are raised on a smaller scale (Nelson et al., 1978). The fish species that are currently available for restocking are presented in Exhibit 2.18 (American Fisheries Society, 1992). The number of fish available for each species and the geographical distribution of the hatcheries are not determinable.

Special equipment (e.g., insulated tank truck with mechanical refrigeration) may need to be rented, leased, or acquired to effectively transport the fish from a hatchery to the point of release (Nelson et al., 1978). A similar type of truck was used to transport the fish from the lake trout hatcheries to the stocking sites in Lake Superior (Great Lakes Fishery Commission, 1987). The fish were then released through pipes connected to the tanks. It is important that the outlet of these pipes or hoses are placed below the surface of the water to reduce the stress on the fish (Smith et al., 1990b).

Constraints

Proper acclimation of the fish between the transporting tank and the point of release is necessary for good survival. Water from the restocking area is slowly pumped into the transportation tank while the transporting water is slowly let out. This acclimation process to temperature, pH, alkalinity, hardness, and salinity alleviates significant stress to the fish. The difference in temperature between the two water types is the prime determinant of the time required. This process should be executed at a rate of at least one hour for every four degrees Celsius in temperature difference (Smith et al., 1990b).

Another consideration for the availability of fish for restocking is the location of the restoration site in relation to the nearest hatchery that raises the same type of fish needed for restocking. If the types of fish injured by contamination are not currently being raised in a hatchery or the nearest hatchery is beyond feasible transportation distance, then a hatchery could be created to raise the type of species needed to restore the injured fish habitat. Two primary limitations exist for creating a new hatchery, adequate clean water supply that is between 50 and 80 degrees Fahrenheit (i.e., depending on whether the species prefers cold or warm water), and the ability to meet current wastewater effluent standards (Nelson et al., 1978).

Exhibit 2.18 Freshwater and marine species available from hatcheries.

Order	Family	Species
ACIPENSERIFORMES	Acipenseridae (Sturgeons)	Acipenser oxyrhynchus (Atlantic sturgeon)
		Acipenser medirostris (Green sturgeon)
		Acipenser fulvescens (Lake sturgeon)
		Scaphirhynchus albus (Pallid sturgeon)
		Acipenser brevirostrum (Shortnose sturgeon)
		Scaphirhynchus platorynchus (Shovelnose sturgeon)
		Acipenser transmontanus (White sturgeon)
	Polyodontidae (Padddlefish)	Polyodon spathula (Paddlefish)
LEPISOSTEIFORMES	Lepisosteidae (Gars)	Atractosteus spatual (Alligator gar)
		Lepisosteus platyrhincus (Florida gar)
		Lepisosteus osseus (Longnose gar)
		Lepisosteus platostomus (Shortnose gar)
		Lepisosteus oculatus (Spotted gar)
AMIIFORMES	Amiidae (Bowfin)	Amia calva (Bowfin)
ANGUILLIFORMES	Anguillidae (Freshwater eels)	Anguilla rostrata (American eel)
OSTEOGLOSSIFORMES	Hiodonidae (Mooneyes)	Hiodon alosoides (Goldeye)
		Hiodon tergisus (Moodeye)
SALMONIFORMES	Salmonidae (Trouts)	Salmo salar (Atlantic salmon)
		Oncorhynchus tshawytscha (Chinook salmon)
		Oncorhynchus keta (Chum salmon)
		Oncorhynchus kisutch (Coho salmon)
		Oncorhynchus gorbuscha (Pink salmon)
		Oncorhynchus nerka (Sockeye salmon)
		Salvelinus alpinus (Arctic char)
		Thymallus articus (Arctic grayling)
		Coregonus spp. (Cisco)
		Salvelinus fontinalis (Brook trout)
		Salmo trutta(Brown trout)
		Oncorhynchus clarki (Cutthroat trout)
		Salvelinus namaycush (Lake trout)
		Prosopium spp. (Whitefish)
		Oncorhynchus mykiss (Rainbow trout)
	Umbridae (Mudminnows)	Umbra spp. (Mudminnow)
	Esocidae (Pikes)	Esox niger (Chain pickerel)
		Esox americanus vermiculatus (Grass pickerel)
		Esox lucius (Northern pike)
		Esox americanus americanus (Redfin pickerel)
		Esox masquinongy (Muskellunge)
		Esox lucius/masquinongy (Tiger muskellunge)

CYPRINIFORMES	Characidae (Characins)	Astynanax mexicanus (Mexican tetra)
	Cyprinidae (Minnows and Carps)	Cyprinus carpio (Common carp)
		Campostoma spp. (Stoneroller)
		Pimephales promelas (Fathead minnow)
		Notemigonus crysoleucas (Golden shiner)
		Ctenopharyngodon idella (Grass carp)
		Other cyprinids
		Ictiobus cyprinellus (Bigmouth buffalo)
		Ictiobus niger (Black buffalo)
		Ictiobus babalus (Smallmouth buffalo)
		Hypentelium etowanum (Alabama hog sucker)
		Moxostoma duquesnei (Black redhorse)
		Moxostoma poecilurim (Blacktail redhorse)
		Cycleptus elongatus (Blue sucker)
		Erimyzon oblongus (Creek chubsucker)
		Moxostoma erythrurum (Golden redhorse)
		Erimyzom sucetta (Lake chubsucker)
		Catostomus catostomus (Longnosre sucker)
		Catostomus platrhynchus (Mountain sucker)
		Hypentelium nigricans (Northern hog sucker)
		Moxostoma macrolepidotum (Shorthead redhorse)
		Moxostoma anisurum (Silver redhorse)
		Catostomus commersoni (White sucker)
		Carpiodes cyprinus (Quillback)
		Carpiodes carpio (River carpsucker)
SILURIFORMES	Ictaluridae (Freshwater catfish)	Ictalurus furcatus (Blue catfish)
		Ictalurus punctatus (Channel catfish)
		Pylodictus olivaris (Flathead catfish)
		Ictalurus catus (White catfish)
		Ictalurus melas (Black bullhead)
		Ictalurus nebulosus (Brown bullhead)
		Ictalurus platycephalus (Flat bullheadd)
		Norurus spp. (Maddtoms)
		Ictalurus natalis (Yellow bullhead)
		Aphredoderus sayanus (Pirate perch)
		Percopsis Omiscomaycus (Trout-perch)
ANTHERINIFORMES	Cyprinidonitae (Killifishes)	Fundulus spp. (Killifish, topminnows, studfish)
	Poeciliidae (Livebearers)	Gambusia affinis (Mosquitofish)
	Atheriniddae (Silversides)	Labidesthes sicculus (Brook silverside)
		Menidia beryllina (Inland silverside)
		Menidia extensa (Waccamaw silverside)
GASTEROSTEIFORMES	Gasterosteidae (Sticklebacks)	Apeltes quaddracus (Fourspine stickleback)
		Gasterosteus aculeatus (Threespine stickleback)
PERCIFORMES	Percichthyidae (Temperate basses)	Morone saxatilis (Striped bass)
		Morone chrysops (white bass)
		Morone mississippiensis (Yellow bass)
		Monone americana (White perch)
	Centrarchiddae (Sunfishes)	Micropterus salmoides (Largemouth bass)
		Micropterus coosae (Redeye bass)
		Micropterus punctulatus (Spotted bass)

		Micropterus dolomieui (Smallmouth bass)
		Pomoxis nigromaculatus (Blackcrappie)
		Pomoxis annularis (White crappie)
		Felassoma zonatum (Banded pygmy sunfish)
		Enneacanthus obesus (Banded sunfish)
		Lepomis macrochirus (Bluegill)
		Enneacanthus gloriosus (Bluespotted sunfish)
		Lepomis marginatus (Dollar sunfish)
		Centrarchus macropterus (Flier)
		Lepomis cyanellus (Green sunfish)
		Lepomis megalotis (Longear sunfish)
		Lepomis humilis (Orangespotted sunfish)
		Lepomis gibbosus (Pumpkinseed)
		Lepomis auritus (Redbreast sunfish)
		Lepomis microlophus (Redear sunfish)
		Ambloplites rupestris (Rock bass)
		Amblopolites ariommus (Shadow bass)
		Lepomis punctatus (Spotted sunfish)
		Lepomis gulosus (Warmouth)
		Perca flavenscens (Yellow perch)
		Etheostoma spp.; Percina spp. (Darters)
		Stizostedion canadense (Sauger)
		Stizostedion vitreum vitreum (Walleye)
		Aplodinotus grunniens (Freshwater drum)
	Cichlidae (Cichlids)	Tilapia melanotheron (Blackchin tilapia)
		Tilapia aurea (Blue tilapia)
		Tilapia mossambica (Mozambique tilapia)
		Tilapia zilli (Redbelly tilapia)
		Tilapia mariae (Spotted tilapia)
	Cottidae (Sculpins)	Cottus spp. (Sculpin)
Source: American Fisheries Society, 1990.		

Future Restoration Actions

In some cases, mortality among restocked fish can be significant. Monitoring the restoration site for the initial two or three days after restocking is important to evaluate survival (Nelson et al., 1978; Smith et al., 1990b). If the mortality rate is higher than 5 percent, then additional restocking is necessary (Smith et al., 1990b).

2.3.2.3.3 Habitat Restoration and Enhancement

See Section 3.3 for a more detailed description of this action. Also, the habitat restoration actions in Section 2.2 apply here as well.

Availability of Services, Materials and Equipment

For reliable information on artificial reef design, development, materials, etc., Duedall and Champ (1991) recommend contacting the Artificial Reef Development Center, a branch of the Sport Fishing Institute. In addition, new developments are discussed periodically at several national and international conferences focused on artificial reefs. Other groups participating actively in artificial reef programs include the federal government, state governments (e.g., California, Florida, North Carolina, Washington), local governments, academic entities, and private companies. NMFS provides guidance through the National Artificial Reef Plan (Duedall and Champ, 1991).

The materials that are feasible to use in the formation of artificial reefs are immeasurable. The type of material can range from readily available items (e.g., old automobile tires) to reef structures constructed specifically for this purpose (e.g., plastic resin formed into a cone shape). Following are several examples of materials used to construct the artificial reefs discussed in the literature:

- According to Duedall and Champ (1991), common materials used internationally for artificial reef construction include aircraft; automobiles, buses, and trolleys, bamboo and bamboo combined with tires; baled garbage; bridges; concrete blocks; construction rubble (concrete debris such as culverts, pile cutoffs); engines; fiberglass and reinforced plastic; freight trains and wheels; metal (primarily steel and iron); quarry rock (i.e., granite, sandstone, limestone); offshore oil and gas platforms; polypropylene rope and cable; polyvinyl chloride piping; refrigerators, stoves, water heaters, and washing machines; ships and boats; stabilized ash (i.e., coal ash, oil ash, incineration ash) in a concrete matrix; sinks and toilets; tires; weapons of war; and wood, trees, and brush. In the U.S., reef engineers are now discouraged from using trash and debris in their designs because of the public perception of dumping instead of reef building and the possibility of contamination and pollution from the debris. Instead, many designs are created with various configurations and combinations of concrete, quarry rock, wood, and tires;

- Feigenbaum et al. (1989) experimented with five reef structure types, unballasted tire bundles, high surface area tires, tires embedded in concrete, concrete igloos, and concrete pipe pyramids. Hueckel et al. (1989) used quarry rock to construct the rocky habitat artificial reefs because of its durable qualities and the large quantities readily available in Washington. The artificial reef development program, discussed by Prince and Maughan (1978), used triangle tire units for the reefs;

- Nelson et al. (1978) examined studies that used brush shelters, tire shelters, and other fish shelters (e.g., rubble, concrete pipe, cement blocks, quarry stone, old cars) for artificial reefs. One experiment in California consisted of creating artificial kelp beds to enhance fishery habitats by placing plastic strips weighted on one end into an appropriate habitat;

- The eight manufactured artificial reef structures evaluated by Bell et al. (1989) consisted of steel-reinforced concrete pipes with holes, larger steel-reinforced concrete pipes, polyolefin plastic cones, polyolefin plastic hemispheres, structural steel cubes, modified structural steel cubes with plastic mesh, modified concrete and PVC docks, and tires embedded in concrete;

- The artificial reef proposed for offsite habitat mitigation of a landfill development project for the Port of Long Beach, described by Knatz (1987), consisted of contaminant-free concrete, rubble, and riprap rock. The rocks were a minimum of 1 foot in diameter and were placed into piles 10 feet high; and

- Obsolete petroleum platforms are another source for artificial reef structure material. This process of converting an unused platform into an artificial reef structure, instead of destroying it, is the rigs-to-reefs concept (Iudicello, 1989; McGurrin and Fedler, 1989).

Similar to artificial reefs, which are usually placed in lakes, oceans, or bays, artificial stream structures can be constructed from various types of material. The artificial stream structures evaluated and studied by Frissell and Nawa (1992) included lateral log deflectors, diagonal log deflectors, cross-stream log weirs, multiple-log structures, cabled natural woody debris jams, and single and clustered boulders. One proposed *Exxon Valdez* restoration project, directed at the restoration of chum salmon habitat and population, involves the installation of instream structures consisting of large boulders and logs (Exxon Valdez Oil Spill Trustees, 1992a).

Nelson et al. (1978) evaluate several types of artificial stream structures used to enhance fish habitats through diversification. Current deflectors are installed in a stream to control and regulate stream flows to benefit fish habitats and decrease bank erosion. There are many current deflector shapes including the triangular wing, peninsular wing, and peninsular wing with chute. These deflectors are constructed from common, natural materials, such as logs, rock, or gabions (wire baskets filled with rocks).

Wesche (1985) and Gore et al. (1988) recommend several types of artificial stream structures that will potentially enhance fishery habitats. These structures include: current deflectors constructed from various formations of logs, rocks, gabions, and wire mesh; low-profile dams constructed from rocks, boulders, logs, and gabions; and single or groups of boulders. The introduction of beaver populations into a suitable habitat is one natural action recommended for the establishment of a low-profile dam structure.

The fish passageways constructed for the Northwest hydroelectric dam-related program consisted of installation of fish ladders and placement of submerged screens blocking the entrance to the turbines. The screens encourage the fish to travel through a chute where the fish will either be released into the river below the dam or loaded onto a barge and released further down the river. During the time period prior to construction of mechanical passageways, two methods of allowing the fish to bypass the dam are to intentionally discharge water over the edge of the dam or to raise the emergency headgates on the dam. Of course, these methods only work for fish migrating toward the ocean. Fish ladders allow movement back up the river (Smallowitz, 1989).

Nelson et al. (1978) evaluate several fish passageway improvement actions, including trap and haul systems, fishways, conduits, culverts, and turbine bypasses. The trap and haul systems are developed to transport migrating fish species through an obstruction (e.g., hydroelectric dam), typically upstream. The fishways evaluated include non-mechanical methods of allowing the fish to swim upstream, such as fish ladders or fish passes. These actions were primarily used to improve the passageway of fish through or around dams. Three types of fish ladders are evaluated, including pool/weir, pool/orifice, and vertical slot ladders. Conduits and culverts are structures established as bypass systems, for both upstream and downstream-migrating fish, around dams and other obstructions. Turbine bypasses are constructed to deter fish travelling downstream from passing through the hydroelectric turbines of dams.

One proposed method to facilitate the restoration of pink salmon populations injured by the *Exxon Valdez* oil discharge is the installation of several fish passageway barrier bypasses on streams important to the pink salmon fish species. The bypasses would consist of channels and steeppasses, which would be anchored with cable for stability. Water diversion structures constructed from gabions reinforced with steel pipe would force water through the channels and steeppasses (Exxon Valdez Oil Spill Trustees, 1992a).

During the construction and installation phases of artificial reef development, special equipment (e.g., crane) may need to be rented or leased. The Smith Mountain Lake artificial reef development program, discussed by Prince and Maughan (1978), used the following pieces of common equipment to construct and deploy the artificial reef structures, crane, barge, tug boat, forklift, tractor, and tractor trailer.

Bell et al. (1989) provide a list of equipment used to deploy the eight types of manufactured artificial reefs evaluated in their study. The reef units were either dropped by a forklift, pushed or rolled in by hand, sunk by swimmers, or sunk and anchored by divers. The structures were deployed from common vessels such as a 30.5 meter research vessel, a 12.2 meter sport fishing boat, and a 33 meter deck-barge and tugboat. In two cases, the structures were towed by a 15.2 meter research vessel. To load the structures on the vessels, a 0.9 metric ton forklift, a 1.8 metric ton forklift, and a 9.1 metric ton crane were used. For the structures constructed from plastic, no additional equipment was needed to load them onto the deployment vessel.

The equipment required for the methods of distribution of limestone examined for the Nova Scotia river liming project includes trucks, tractors, boats, helicopters, or airplanes capable of distributing limestone, and various road construction equipment (Watt, 1986).

Constraints

Although many different types of materials may be used to construct an artificial reef, there are several factors to consider, besides availability and short-term cost effectiveness, when selecting appropriate material. Hueckel et al. (1989) stress the importance of using durable material for the construction of artificial reefs. Fragile substances will deteriorate at a rate that will require frequent repairs or replacement, thus causing unnecessary disturbance to the habitat. The ideal situation is to use the most durable material that is also readily available and cost effective, such as the quarry rock they used to mitigate a rocky subtidal habitat.

Another consideration discussed by Hueckel et al. (1989) is the selection of an appropriate reef site. Their major concern was disturbance from vessel traffic and commercial net fisheries.

Feigenbaum et al. (1989) indicate a variation on structural stability and mobility considerations based on the location of the reef. Their study found that reefs placed in coastal waters were less stable and more mobile in coastal waters than in protected or semi-protected waters (e.g., the Chesapeake Bay), mostly due to storm activity.

Duedall and Champ (1991) provide a comprehensive list of other factors to consider before selecting an artificial reef site. These considerations include accessibility to and distance from shore; availability of reef-building materials; biological characteristics of the site and adjacent areas; depth of photic zone; detriments (i.e., vessel lanes); ease of reef deployment; liability, insurance, and permit requirements; oceanographic characteristics, currents, and wave conditions; projected uses and benefits of the site, both economic and recreational; sedimentation rate; target species; turbidity; and weather and storms.

McGurrin and Fedler (1989) provide several issues to consider during the siting phase of an artificial reef in the rigs-to-reefs program. These considerations follow general coastal zone mapping procedures and include assessment of the current marine recreational fishing industry, location of the important recreational fishing zones, and elimination of the areas with potential interference to the artificial reef activities (i.e., shipping lanes, military warning zones, and marine sanctuaries).

Artificial stream structures are not recommended for installation in streams where the gradient exceeds 3 percent or where the stream flow fluctuates substantially, according to the U.S. Forest Service (Nelson et al., 1978). The exception to this guideline is the low-profile dam structure. Dams can be effective up to a 20 percent gradient level. In addition, current deflectors and dams should not protrude more than 0.3 meters above the low-flow level. The deflectors should also be angled downstream at about a 45 degree angle from the current (Wesche, 1985).

According to the U.S. Forest Service, it is recommended that brush shelters, which are constructed in various forms from brush and trees, be placed in an area with approximately four meters of water and weed-free, hard bottoms. If more than one shelter, or artificial reef, is installed in an area, they should be separated by at least 45 meters (Nelson et al., 1978).

It is also recommended that the design for a fish ladder include drops no longer than 30 centimeters. The orifices should be no larger than 1.2 square meters on the pool and orifice ladder. The overall vertical height of any fish ladder should be 30 meters or less (Nelson et al., 1978).

There are operational limitations related to many of the fish passageway improvement methods. For all fish passageways, the opening must be easily accessible and attractive to fish. An operational constraint related to the trap and haul system, used to transport fish upstream, is the potential for injury of the fish. In some cases, the trap and haul system is the combination of a fish ladder and hopper shaft. However, this system could also consist of trapping the fish in a barge, transporting them to a new location, and releasing the fish. The latter method has a higher potential for injury to the fish. Debris accumulates easily in the pool/weir or pool/orifice fish ladders. These ladders also can not tolerate large shifts in water levels. The vertical slot ladder does not have the debris problem works more effectively when the water levels are equal at both ends (Nelson et al., 1978).

2.3.2.3.4 Modification of Management Practices

Intensified monitoring and management of fishery stocks (especially coastal cutthroat trout, pink salmon, sockeye salmon, Pacific herring, rockfish, and Dolly Varden) was proposed for several related restoration projects following the *Exxon Valdez* oil discharge. This increase in fishery management typically includes shifting recreational and commercial fishing efforts away from injured stocks to alternative sites that were not affected by the discharge (Exxon Valdez Oil Spill Trustees, 1992a,b).

Prior to establishing management policies related to fisheries use, a database with population, size, and other vital information about each fishery at various sites should be developed and maintained. Acquisition of these data would require intensive field work (Exxon Valdez Oil Spill Trustees, 1992a).

2.3.2.3.5 Habitat Protection and Acquisition

Under consideration by the Exxon Valdez Oil Spill Trustees (1992b) are two fishery habitat protection and acquisition strategies. The first plan includes the designation of specific injured regions as protected marine habitat areas (i.e., national marine sanctuaries, marine parks). The second proposal under consideration is the acquisition of private areas for the purpose of recreational fishing. This would alleviate the pressure on recovering sport fishing stocks. The applicability of such alternatives is highly site-specific and depends on the availability of appropriate lands in a particular region.

2.3.3 Reptiles

There exist three technically-feasible actions for restoring injured reptile populations. These actions include:

- Natural Recovery;
- Restocking/Replacement; and
- Protection of Nest Sites.

Protection of nest sites requires the development and implementation of measures to secure and preserve the sites from predators, human interference, beach erosion, pollution, and other forms of perturbation.

2.3.3.1 Oil Related Literature

The text "Restoration of Habitats Impacted by Oil Spills," edited by Cairns and Buikema (1984), includes information on the restoration of sea turtles injured or destroyed by an oil discharge. The suggested restoration method is restocking, using an alternate site if full restoration of the discharge site is unattainable, with hatchery-reared turtles.

2.3.3.2 Non-oil Related Literature

A report developed by International Animal Exchange, Inc. (1992), an international company specializing in animal procurement and relocation (for zoos and aquariums), provides the availability and cost estimates to deliver live wildlife specimens from captive sources for the purpose of reintroduction to the wild in U.S. territories and the cost estimates to obtain, transport, and acclimate wildlife specimens from the wild. The availability of relocating wild species from other locations to the affected area depends on the terms of the permit acquired for such an activity. Additional information on the actual process of relocation or replenishment of a wildlife population and the predicted survival rates from such activities was obtained through personal communication (Hunt, 1993).

Two similar studies, prepared by the Loggerhead/Green Turtle Recovery Team for the National Marine Fisheries Service (NMFS) and the Southeast Region of the U.S. Fish and Wildlife Service, describe the proposed recovery plans for the Loggerhead and Atlantic Green turtles (NMFS, 1990a,b). In addition to describing the objectives and outline of the recovery plans, these studies also describe the population characteristics, distribution, and size, threats to the turtle nesting and marine environments, and conservation accomplishments in the nesting and marine environments. Sea turtle restoration plans are discussed fully in Section 3.3.3.

2.3.3.3 Technical Feasibility of Restoration Actions

The following subsections discuss the technical feasibility of three actions for reptile restoration.

2.3.3.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for discussion of recovery.

2.3.3.3.2 Restocking/Replacement

Restocking entails either relocating the necessary species to the restoration area from another location or supplementing the injured reptile population with captive raised species. This restocking/replacement action is evaluated below.

Availability of Services, Materials, and Equipment

In addition to the International Animal Exchange, Inc. described above, there are several companies that conduct similar animal procurement and relocation operations. All of these companies are members of the American Association of Zoological Parks and Aquariums (AAZPA). The names, addresses, and telephone numbers of these companies are listed below (AAZPA, 1990):

- Fauna Research & Development, Inc.
  Bard Avenue
  Red Hook, NY 12571
  (914) 758-2549

- International Animal Exchange, Inc.
  E. Nine Mile Road
  Ferndale, MI 48220
  (313) 398-6533

- International Zoological Distributors
  Herve Beaudry
  Laval, P., Quebec, Canada H7E 2X6
  (514) 661-8081

- Lamkin Wildlife Company
  Box 5843
  Amarillo, TX 79117
  (806) 383-4085

- Nelson's Twin Oaks Farm
  Bethany Road
  Alpharetta, GA 30201
  (404) 475-4918

- Earl Tatum
  Pleasant Ridge Drive
  Eureka Springs, AR 72632
  (501) 253-9696

- Zeehandelaar, Inc.
  Sickles Avenue
  New Rochelle, NY 10801
  (914) 636-2096

- Zoological Animal Exchange
  Route 610, Box 164
  Natural Bridge, VA 24578
  291-3205

Some of these companies have experience in all types of wildlife, while others concentrate on only a few types of species. Therefore, during an actual restoration project where many different species are involved, it may be necessary to acquire the services of more than one company.

These firms have expertise in the process of wildlife acclimation and transportation. They also typically own or have access to the proper equipment necessary for successful transportation of the species to the restoration site and acclimation of the species into their new habitat (Hunt, 1993).

The estimated quantities of captive raised or intensively managed reptiles available for restocking purposes are provided in Exhibit 2.19. If the required number of animals are not available from captivity, then the remainder could be relocated from the wild. Relocation of reptiles is typically not feasible or permitted in the United States (Hunt, 1993) and consideration of impacts on the donor population must be made.

Constraints

The primary operational constraints associated with restocking reptile populations are logistics (e.g., the population being restocked is difficult to reach by humans), procurement of required permits, climate, finance, and public interference (Hunt, 1993).

Prior to any restocking or relocating activities, the animal supplier should conduct an in depth study of the species involved. An optimum age for each particular species should be determined. Typically, a juvenile of the species is selected as the most adaptable lifestage. The juvenile is usually the least susceptible to stress from translocation because species at this age are psychologically and physically more adaptable. Restocking, translocating a species from captivity to the wild, has a higher impact on the stress level of an animal than relocating the species from one wild habitat to another (Hunt, 1993).

Future Restoration Actions

The period of time a newly acclimated population is monitored following a relocation or restocking activity depends on the circumstances of the situation. For some populations (e.g., a sea turtle or migratory bird population), monitoring is difficult or not feasible. In other cases, where the populations are gradually acclimated to the wild, monitoring and support of the population is required throughout the transition period sometimes continuing for several generations (Hunt, 1993).

It is expected that some mortalities will occur after translocation of a population. These mortality rates, however, are difficult to estimate for even generic classes of wildlife species. The expected mortality rates include many factors that are specific to the situation and species involved. Any mortalities experienced after a relocation or restocking effort are not covered by the service provider. In a few cases, third party insurance was obtained to meet specific contractual requirements, but this is not standard practice (Hunt, 1993).

Exhibit 2.19 Availability of captive raised reptiles for restocking purposes.

Family	Species	Number of Reptiles Available
Cheloniidae	Atlantic loggerhead turtle	0
	Pacific loggerhead turtle	0
	Atlantic ridley turtle	0
	Pacific ridley turtle	0
Dermochelyidae	Atlantic leatherback turtle	0
	Pacific leatherback turtle	0
Crocodylidae alligatorinae	American alligator	5,000

Source: International Animal Exchange, 1992.

2.3.3.3.3 Protection of Nest Sites

There are many measures recommended in the recovery plans of the Atlantic Green and Loggerhead turtles, that are technically feasible to implement for protection of the nesting habitats of sea turtles (NMFS, 1990a,b). These measures include:

- Developing predator control programs;
- Controlling beach nourishment process;
- Preventing degradation of nesting sites from beach/shoreline erosion control measures;
- Enhancing nesting habitats;
- Acquiring/protecting important nesting beaches;
- Removing exotic vegetation; and
- Protecting nesting habitats from human interference (e.g., artificial lights, foot/vehicular traffic, poaching) through ordinances, regulations, and educational materials.

Constraints

The preferable method of protecting nest habitats involves a minimum of disturbance to the nesting population with a maximum of effectiveness in preventing injury of the nest sites. Nests are relocated only in situations where no other alternatives exist. Artificial incubation of turtle eggs is typically avoided. Most government agencies strive for implementing protective measures that yield a 50 percent hatch rate (NMFS, 1990a,b).

A majority of the protection measures listed above require a high level of cooperation between federal, state, and local officials. Effective monitoring of each situation prior to the implementation of protection measures is an essential phase of the process. Government agencies are needed to implement the control measures of the beach nourishment and beach/shoreline erosion control processes and develop and enforce ordinances and regulations that control human interference with the nest habitats. The involvement of government agencies and non-profit organizations is also necessary for the development and distribution of educational material to increase the public awareness of injury to the nesting sites which results from certain human activities.

2.3.4 Birds

Five general alternatives for restoring injured bird populations include:

- Natural Recovery;
- Restocking/Replacement;
- Habitat Restoration and Enhancement;
- Modification of Management Practices; and
- Habitat Protection and Acquisition.

2.3.4.1 Oil Related Literature

The "1993 Draft Work Plan" and comprehensive 1992 preliminary restoration plan for the *Exxon Valdez*, described in detail in Section 2.3.2.1, describe several restoration alternatives for bird populations affected by the oil discharge (Exxon Valdez Oil Spill Trustees, 1992a, b). These alternatives include reducing human disturbance at bird colonies, controlling harvest of sea ducks, and eliminating continuous oil contamination of prey substrates.

2.3.4.2 Non-oil Related Literature

As described in Section 2.3.3.2, International Animal Exchange, Inc. (1992) developed a report that provides availability levels for stocking of various wildlife species. A majority of the species included are birds. The availability of captive-raised birds and the technical feasibility of restocking are discussed below.

2.3.4.3 Technical Feasibility of Restoration Actions

The following subsections discuss the technical feasibility of restoration actions for bird populations.

2.3.4.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for discussion of recovery.

2.3.4.3.2 Restocking/Replacement

A discussion on the technical feasibility of the wildlife restocking/replacement restoration action is located in Section 2.3.3.3.2.

Availability of Services, Materials, and Equipment

Refer to this subheading in Section 2.3.3.3.2. for a discussion on the availability of restocking/replacement services for wildlife restoration purposes. The estimated quantities of captive raised birds available for restocking purposes are provided in Exhibit 2.20.

There are several significant considerations associated with the relocation of birds. In addition to the issues referred to in Section 2.3.3.3.2, the restoration facilitators need to ensure that the species taken from one population and relocated to a new site do not cause adverse effects on the original population. Although it varies by species, bird populations can normally withstand a loss of 2 to 6 percent (Hunt, 1993).

Constraints

Refer to this subheading in Section 2.3.3.3.2. for a discussion on the various operational constraints related to restocking.

Future Restoration Actions

Refer to this subheading in Section 2.3.3.3.2. for a discussion on the need and capability of future restoration actions after restocking.

2.3.4.3.3 Habitat Restoration and Enhancement

Nelson et al. (1978) recommend two feasible types of bird habitat restoration and enhancement actions: construction of artificial nesting structures and man-made nesting islands. The nesting structures are appropriate for ducks, geese, cormorants, eagles, ospreys, herons, and other species. The nesting islands are suitable for migrating bird species and nesting waterfowl and shorebirds.

Availability of Services, Materials, and Equipment

The nesting structures are typically constructed from wood or metal, although wood is preferable. Nesting islands are developed from both gravel and dredge spoil. Nest enclosures on the islands are constructed from natural materials (e.g., driftwood). Nesting materials, which should be replaced annually, can consist of wild hay, straw, or wood shavings (Nelson et al., 1978).

Exhibit 2.20 Availability of captive raised birds for restocking purposes.

Family	Species	Number of Birds Available
Gaviidae	Common loon	0
Podicipedidae	Horned grebe	0
	Red-necked grebe	0
Domedeidae	Laysan albatross	0
	Black-forested albatross	0
Procellariidae	Northern fulmar	0
	Japanese petrel	0
	Hawaiian petrel	0
	Greater shearwater	0
	Sooty shearwater	0
	Manx shearwater	0
	Short-tailed shearwater	0
Hydrobatidae	Least storm petrel	0
	White-vented storm petrel	0
	Band-rumped storm petrel	0
	Ashy storm petrel	0
	Ringed storm petrel	0
	Leaches storm petrel	0
Pelecanidae	American white pelican	10
	Brown pelican	300
Sulidae	Northern gannet	0
	Blue-footed booby	0
Phalacrocoracidae	Double crested cormorant	0
	SW Double-created cormorant	0
	NW Double-created comorant	0
	Common (great) cormorant	400
	Northern great cormorant	0
	Olivaceous cormorant	0
Ardeidae	American bittern	0
	Great blue heron	10
	Green heron	0
	Tricolored heron	0
	Black-crowned night heron	0
	Night heron	0
	Yellow-crowned night heron	0
	Cattle egret	900

Family	Species	Number of Birds Available
	Snowy egret	100
Threskiornithidae	American white ibis	100
	Scarlet ibis	100
	Bare-faced ibis	0
	White-faced ibis	0
	Glossy ibis	800
	Roseate spoonbill	40
Phoenicopteridae	American flamingo	100
Anatidae	White-fronted goose	30
	Tule goose	0
	Graying goose	30
	Snow goose	100
	Greater snow goose	0
	Lesser snow goose	50
	Emperor goose	30
	Ross goose	30
	Lawrences brant goose	0
	Pacific brant goose	0
	Canada goose (generic)	1,000
	Whistling swan	0
	Trumpeter swan	240
	Duck (most species; generic)	500
Accipitridae	Hawk/Eagle (most species; generic)	0
Gruidae	Whooping crane	0
	Sandhill crane	50
	Lesser sandhill crane	50
	Florida sandhill crane	50
	Mississippi sandhill crane	0
	Canadian sandhill crane	0
	Greater sandhill crane	0
Aramidae	Limpkin	0
Rallidae	Rail/Coot (most species; generic)	0
Haematopodidae	American oystercatcher	0
Recurvirostridae	Hawaiian stilt	0
	Black-winged stilt	0
	Black-necked stilt	0
	American avocet	0
Charadrilidae	Lesser golden plover	0
	Black-bellied plover	0

Family	Species	Number of Birds Available
Scolopacidae	Spotted sandpiper	0
	Upland sandpiper	0
	Willet	0
	Wandering tattler	0
	Godwit	0
	Long-billed curlew	0
	Lesser yellowlegs	0
	Greater yellowlegs	0
	Solitary sandpiper	0
	Black turnstone	0
	Andean snipe	0
Laridae	Gull/tern (most species; generic)	0
Alcidae	Puffin (most species; generic)	10

Source: International Animal Exchange, 1992.

Constraints

A significant constraint on the design and siting of both the nest structures and islands is the protection from predators. This can easily be achieved by installing a fence around the site or positioning the nest several feet or more off the ground or water. Another consideration for placement of a nest on or near water is the fluctuation in flow or the flood level of water. This fluctuation should be controlled as much as possible during the nesting season (Nelson et al., 1978).

This restoration action should be considered temporary in most cases. It is designed to provide nesting shelter until more permanent, natural facilities are reestablished in the habitat (Nelson et al., 1978).

2.3.4.3.4 Modification of Management Practices

The Exxon Valdez Oil Spill Trustees have proposed to reduce disturbance to bird colonies (i.e., specifically the common murres) to allow the restoration process to continue free from human disturbance. This includes educating appropriate industries (e.g., commercial fishing) of the methods proposed to reduce disturbance and to establish strict enforcement of the Migratory Bird Treaty Act. Modification of fishing gear (e.g., gillnets) or fishing practices could protect diving seabirds such as marbled murrelets. The Exxon Valdez Trustees are also considering restrictions on the legal harvest of sea ducks by shortening the length of the hunting season and reducing bag limits (Exxon Valdez Oil Spill Trustees, 1992a,b).

In addition to protection from disturbance and hunting as effective management practices for restoration of seabirds, Nur and Ainley (1992) suggest the protection of prey availability through monitoring and controlling fisheries important to the seabird species. The feasibility of this alternative is not documented.

2.3.4.3.5 Habitat Protection and Acquisition

Designating injured bird habitats and implementing and expanding buffer zones are possible actions for habitat protection and acquisition. These actions were recommended by the Exxon Valdez Oil Spill Trustees (1992b) in protecting marine areas, and creating nesting areas for seabirds, sea ducks, and bald eagles.

2.3.5 Mammals

Five actions for restoring injured mammal populations are:

- Natural Recovery;
- Restocking/Replacement;
- Habitat Restoration and Enhancement;
- Modification of Management Practices; and
- Habitat Protection and Acquisition.

2.3.5.1 Oil Related Literature

The "1993 Draft Work Plan" and the comprehensive 1992 preliminary restoration plan for the *Exxon Valdez* suggest several restoration actions for mammal populations affected by the oil discharge (Exxon Valdez Oil Spill Trustees, 1992a and 1992b). These actions include reducing human disturbance at marine mammal haul-out sites, controlling harvest of specific marine and terrestrial mammals, and eliminating continuous oil contamination of prey substrates.

Cairns and Buikema (1984) provide information on the restoration of marine mammals injured by an oil discharge. One restoration action, suggested for implementation, is restocking the restored habitat or an alternative site if full restoration of the discharge site is unattainable.

2.3.5.2 Non-oil Related Literature

As described in Section 2.3.3.2., International Animal Exchange, Inc. (1992) reports the availability of various wildlife species. Several of the species included are marine mammals. The availability of captive-raised mammals and technical feasibility of successfully replenishing an affected mammal population is discussed in detail in the following discussion.

2.3.5.3 Technical Feasibility of Restoration Actions

The following subsections discuss the technical feasibility of restoration actions for mammals.

2.3.5.3.1 Natural Recovery

Monitoring of natural recovery is technically feasible. See Chapter 3 for a discussion of recovery.

2.3.5.3.2 Restocking/Replacement

A discussion on the technical feasibility of the wildlife restocking/replacement restoration action is located in Section 2.3.3.3.2.

Availability of Services, Materials, and Equipment

Refer to Section 2.3.3.3.2. for a discussion on the availability of restocking/replacement services for wildlife restoration purposes. The estimated quantities of captive raised mammals available for restocking purposes are provided in Exhibit 2.21. The suppliers claim high survival rates of these animals, assuming care and effort is taken as indicated by the costs in Chapter 4.

As mentioned in Section 2.3.4.3.2, wildlife populations can withstand a small decrease without adverse effects. Mammal populations can sustain of a loss of approximately 2 to 4 percent. However, this amount does vary by species (Hunt, 1993).

Constraints

Refer to Section 2.3.3.3.2. for a discussion on the various operational constraints related to restocking.

Future Restoration Actions

Refer to Section 2.3.3.3.2. for a discussion on the need and capability of future restoration action after restocking.

2.3.5.3.3 Habitat Restoration and Enhancement

While provision or improvement of appropriate sites for reproductive or feeding activities could be considered, no documentation of their use is available. General habitat enhancement actions could be conducive to mammal population recovery.

Exhibit 2.21 Availability of captive raised mammals for restocking purposes.

Family	Species	Number of Mammals Available
Cricetidae	Muskrat	0
Delphinide	Killer whale	0
	False killer whale	0
	Northern right-whale dolphin	0
	Saddle back dolphin	0
	Common dolphin	0
	Risso's dolphin	0
	White-sided dolphin	0
	Pacific white-sided dolphin	0
	Gill's bottle-nosed dolphin	0
	Bottle-nosed dolphin	10
	Pacific harbour porpoise	0
	Dall's porpoise	0
Monodontidae	Beluga whale	0
Ursidae	Polar bear	30
Mustelidae	Northern sea ottter	10
	Southern sea otter	0
Otariidae	Northern fur seal	100
	Steller's northern sea lion	0
	California sea lion	50
	Walrus	0
	Bearded seal	0
	Grey seal	20
	Harbor seal	40
	Northern elephant seal	0
	Hawaiian monk seal	0
Trichechidae	Manatee	0

Source: International Animal Exchange, 1992.

2.3.5.3.4 Modification of Management Practices

The Exxon Valdez Oil Spill Trustees are considering the implementation of two modifications to current management practices related to mammals. These actions include the reduction of disturbance at marine mammal haul-out sites and the development of alternative harvest guidelines. The issues related to these actions are discussed in Section 2.3.4.3.4. These management practices would be focused on sea otters, harbor seals, river otters, and brown bears. Many restrictions are already established by the Marine Mammal Protection Act, although stricter enforcement of the above act is proposed (Exxon Valdez Oil Spill Trustees, 1992a,b).

Consideration (in Exxon Valdez restoration planning) is also being given to voluntary use of different fishing gear (pot gear in lieu of long time) for black cod and, possibly, Pacific cod and halibut. This would potentially reduce fishery interactions of killer whales, since killer whales have historically raided long lines in Prince William Sound.

Nur and Ainley (1992) recommend the reduction or elimination of commercial harvesting and incidental killing of pinnipeds and cetaceans as the most effective and feasible modification to management practices.

2.3.5.3.5 Habitat Protection and Acquisition

Similar to fishery habitat protection and acquisition, the Exxon Valdez Oil Spill Trustees also are considering the designation of injured marine mammal habitats as protected marine areas (Exxon Valdez Oil Spill Trustees, 1992b). Refer to Section 2.3.2.3.5 for a complete discussion of this action.

2.4 Replacement Actions

The replacement action is used extensively to compensate for oil discharge-related injuries. Some of which are briefly discussed.

The restoration approach in the OPA restorations favors primary restoration. However, from practical, cost-effectiveness, and scientific perspectives, primary restoration is not the implemented restoration strategy in a number of cases. Compensatory alternatives that do not encompass direct resource or habitat restoration and are often referred to as mitigation. Examples of compensatory actions that have been developed for the mitigation of a habitat through acquisition, service enhancement, or protection/management include:

- Habitat Creation;
- Land Protection;
- Public Access Improvements;
- Other Recreational Facility Improvements;
- Habitat Enhancement;
- Resource Management Practices;
- Pollution Control Activities; and
- Public Awareness Activities.

The relationships between these habitat and resource compensatory actions and the habitat types discussed previously in Sections 2.2 and 2.3 are provided in Exhibit 2.22. There are an exhaustive number of compensatory actions at the habitat-specific level.

The most exhaustive exploration of mitigation strategies, and the one described here, was associated with the *Exxon Valdez* efforts to develop mitigation plans for habitats and services injured by the *Exxon Valdez* discharge. One such document by the Exxon Valdez Oil Spill Trustees (1992a) provides a summary of the habitats and species injured and gives brief descriptions of several potential restoration actions for the resources and/or services affected by the discharge. This report does not represent the final and complete restoration plan for the *Exxon Valdez* discharge, but it does represent the most comprehensive oil discharge restoration plan available and addressed a broader range of alternative than previously undertaken. This plan provides descriptions of several proposed projects related to habitat and resource protection. Another report prepared by the *Exxon Valdez* Oil Spill Trustees (1992c) is the "1993 Draft Work Plan" which summarizes the restoration projects currently under consideration. The projects will be completed through a joint effort from various agencies of the federal government and the state of Alaska

Exhibit 2.22 Compensatory restoration actions.

Habitat Types	Habitat Creation	Land Protection	Public Access Improve-ments	Other Recrea-tional Facility Improvements	Habitat Enhancement (Artificial Reefs, Etc.)	Resource Management Practices	Pollution Control Activities	Public Awareness Activities
Estuarine and Marine Wetlands								
Saltmarsh	🗐	🗐	🗐				🗐	🗐
Mangrove	🗐	🗐	🗐				🗐	🗐
Freshwater Wetlands								
Emergent Wetlands	🗐	🗐	🗐				🗐	🗐
Scrub/Shrub Wetlands	🗐	🗐	🗐				🗐	🗐
Forested Wetlands	🗐	🗐	🗐				🗐	🗐
Bogs and Tundra	🗐	🗐	🗐				🗐	🗐
Vegetated Beds								
Macroalgal Beds	🗐	🗐					🗐	
Seagrass Beds	🗐	🗐					🗐	
Freshwater Aquatic Beds	🗐	🗐					🗐	
Mollusc (Oyster) Reefs	🗐	🗐	🗐				🗐	🗐
Coral Reefs	🗐	🗐	🗐		🗐		🗐	🗐
Estuarine and Marine Intertidal								
Intertidal Rocky Shore		🗐	🗐	🗐			🗐	🗐
Intertidal Cobble-Gravel Beach		🗐	🗐	🗐			🗐	🗐
Intertidal Sand Beach		🗐	🗐	🗐			🗐	🗐
Intertidal Mud Flat		🗐					🗐	
Estuarine and Marine Subtidal								
Rock Bottom							🗐	
Cobble-Gravel/Sand/Silt-								

Habitat Types	Habitat Creation	Land Protection	Public Access Improve-ments	Other Recrea-tional Facility Improvement s	Habitat Enhancement (Artificial Reefs, Etc.)	Resource Managemen t Practices	Pollution Control Activities	Public Awareness Activities
Mud Bottom							🗐	
River and Lacustrine Shorelines								
Rock Shore		🗐	🗐	🗐			🗐	🗐
Cobble-Gravel Shore		🗐	🗐	🗐			🗐	🗐
Sand Shore		🗐	🗐	🗐			🗐	🗐
Silt-Mud Shore		🗐					🗐	
Riverrine Bottom								
Rock Bottom							🗐	
Cobble-Gravel/Sand/Silt-Mud Bottom							🗐	
Lacustrine Bottom								
Rock Bottom							🗐	
Cobble-Gravel-Sand/Silt-Mud Bottom							🗐	
Biological Resources								
Shellfish		🗐			🗐	🗐		
Fish		🗐			🗐	🗐		
Reptiles		🗐			🗐	🗐		
Birds		🗐			🗐	🗐		
Mammals		🗐			🗐	🗐		

government. The projects are divided by the following categories for restoration and replacement activities: management action; damage assessment; monitoring; enhancement; technical support; manipulation; habitat protection and acquisition; and land protection.

A prime example of a non-discharge mitigation guidance document is Nelson et al. (1978), a handbook for the U.S. Department of the Interior which summarizes nearly 300 fish and wildlife habitat and population improvement actions. The actions discussed include enhancement actions proven effective during previous dam and reservoir projects or determined to be potentially effective. One section describes the process of land acquisition as a method of habitat restoration and protection.

2.4.1 Technical Feasibility of Replacement Actions

The following paragraphs discuss technically feasible replacement actions and provides examples of each. Again, these mitigation strategies are offered as examples of the range of actions which may be available and is by no means exhaustive.

2.4.1.1 Habitat Creation

After locating a site suitable to sustain a new habitat, actions similar to primary restoration efforts (i.e., grading, planting, supplementary erosion control structures, and sediment removal/replacement) can be used. In general, the strategy should identify a site with the potential of providing an array of critical habitat and natural resource services. This site may be one injured by prior releases of hazardous materials or oil or simply a location in need of environmental enhancement. In some cases the site could even be general land acquired for the specific purpose of habitat creation (e.g., purchase and grading down of upland for saltmarsh creation).

2.4.1.2 Land Protection

Nelson et al. (1978) provides additional information on the protection of wildlife habitats during reservoir and dam projects through land acquisition. They suggest that land can be acquired through purchase, easement, or lease transactions. Based upon these project experiences, a primary constraint is the ability to acquire sufficient land to meet the objectives of the acquisition.

Following the *Exxon Valdez* oil discharge, the need arose for the establishment of protective measures for various non-biological sites. In order to protect the archeological sites and artifacts within the discharge area, which already were vandalized, the Exxon Valdez Oil Spill Trustees implemented a site stewardship program, consisting of a group of local individuals who are to watch remotely-located archeological sites. This program is similar to successful archeological site stewardship programs in Arizona and Texas (Exxon Valdez Oil Spill Trustees, 1992a,b).

The Exxon Valdez Oil Spill trustees are recommending that the oil discharge area be designated a "special management area." This would ensure that any activities requiring permits from the state (e.g., log transfer sites) were not in conflict with the recovery and restoration of injured nataural resources and services. The trustees are also considering that one or more sites should be designated marine protected areas. This designation by the trustee agencies, the Alaska State Legislature, and Congress would help protect the biological natural resources inhabiting the area and preserve the area for recreation and research activities (Exxon Valdez Oil Spill Trustees, 1992b).

Although the State of Alaska and federal governments own a majority of the tidelands that were injured by the discharge, several areas are still owned by municipalities or private individuals. Acquisition by the state of these other areas would provide officially protected habitat for the injured species and create an alternative site for natural resource users. Through easements, property rights, or fee-simple title, the trustees are also investigating the acquisition of upland forests and watersheds within the oil discharge area to ensure protection of vital stream and river areas. Another type of acquisition considered by the trustees is acquiring "inholdings" within existing parks and refuges from willing sellers to further sustain services and provide sufficient refuge for biological natural resources (Exxon Valdez Oil Spill Trustees, 1992b).

2.5 Legal and Regulatory Constraints

Even the most beneficial of restoration actions are subject to a wide variety of legal and regulatory conditions beyond those associated with the damage assessment and restoration planning processes. These influences on restoration actions range from requirements for relatively perfunctory notification, to elaborate multi-agency permitting procedures. As noted in Woodhouse (1979) and Chianelli (1992), these factors have the potential to materially affect the timing and operational feasibility of a project. Because these legal and regulatory factors represent a commonality among many of the restoration actions addressed in this document, they are consolidated into this section.

Exhibit 2.23 Range of federal agency roles potentially affecting implementation of restoration strategy.

FEDERAL AGENCY	SCOPE OF RESOURCE AND HABITAT MANAGEMENT RESPONSIBILITY	LEGISLATIVE AUTHORITY	POTENTIAL PROGRAMS
U.S. Environmental Protection Agency (EPA)	Protect, maintain, restore and enhance water quality	Clean Water Act (P.L. 92-500)). 33 U.S.C. 1251 et seq.	1. National Estuary Program (§320) 2. Discharge permits (NPDES program) (§402)) 3. Oil and hazardous substance spills (§311) 4. Toxic (priority) pollutant and pretreatment program (§307) 5. Nonpoint source control program (§319) 6. Chesapeake Bay program (§117) 7. In-place pollutants (§115) 8. Dredge and fill wetlands program (§404)
	Avoid unreasonable degradation or endangerment of the marine environment or public health	Natioinal Marine Sancturaries Act (P.L. 92-532), 33 U.S.C., 1401 et seq., as amended by the Ocean Dumping Ban Act of 1988 (P.L. 100-688)	1. Site designation of ocean dumpsites for wastes and dredged material [§102(c)] 2. Veto of U.S. Army Corps of Engineers (USACOE) permits for dredged material ocean dumping (§103)
	Regulate pesticide chemicals	Federal Insecticide, Fungicide, and Rodenticide Act (P.L. 92-516), 7. U.S.C. 136 et seq.	Setting of action levels of tolerances for unavoidable pesticide contaminants in fish and shellfish (Food, Drug and Cosmetic Act, §408)
U.S. Department of Transportation (DOT)	Enhance marine life	Reefs for Marine Life Conservation (P.L. 92-402), National Fishing Enhancement Act of 1984 (P.L. 98-623), 16 U.S.C. 1220-1220d	Use of obsolete ships as artificial reefs for the conservation of marine life
	Enforcement of fisheries laws (U.S. Coast Guard)	(Magnuson) Fishery Conservation and Management Act (P.L. 94-265), 16 U.S.C 1801 et seq.	Enforcement of restrictions on commercial fishing within the fishery conservation zone (Exclusive Economic Zone) (§311)
National Oceanic and Atmospheric Administration (NOAA)	Natural resource trustee for: marine fishery resources and supporting ecosystems; anadromous fish; certain endangered species and marine mammals; National Marine Sanctuaries; and Estuarine Research Reserves	Clean Water Act (P.L. 92-500), 33 U.S.C. 1321(f)(5) Comprehensive Environmental Respponse, Compensation, and Liability Act (P.L. 96-510), 42 V.S.C. 9601 et seq, Oil Pollution Act q1990 (P.L. 101-380), 33 V.S.C. 2701 et seq.	1. Remedial Action Program (CERCLA, §104) 2. NRDA (CERCLA, §107) 3. (OPA, §1006)
	Marine mammals	Marine Mammal Protection Act of 1972 (P.L. 9-522), 16 U.S.C. 1361 et seq.	Prohibition or strict regulation of the direct or indirect taking or importation of marine mammals
	Anadromous Fish	Anadromous Fish Conservation Act of 1965 (P.L. 89-304), 16 U.S.C. 757a-757g	Conservation, development, and enhancement of anadromous fishery resources
		Salmon & Steelhead Conservation and Enhancement Act of 1980 (P.L. 96-561) 16 U.S.C. 3301-3345	Management and enhancement of salmon and steelhead stocks
	Threatened and endangered species and their critical habitats	Endangered Species Act of 1973 (P.L. 93-205), 16 U.S.C. 1531 et seq.)	Insurance that any action authorized, funded, or carried out by any Federal agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification

FEDERAL AGENCY	SCOPE OF RESOURCE AND HABITAT MANAGEMENT RESPONSIBILITY	LEGISLATIVE AUTHORITY	POTENTIAL PROGRAMS
			of habitat critical to such species (§7) (covers marine species)
	Marine fisheries	Magnuson Fishery Conservation and Management Act of 1976 (P.L. 94-265) 16 U.S.C. 1801 et seq.	Fishery Management Plans by eight regional Fishery Management Councils
	Marine sanctuaries	Marine, Protection, Research and Sanctuaries Act (Title III) P.L. 92-532), 16 U.S.C. 1431-1439)	National Marine Sanctuaries Program
	Protection of coastal natural resources, including wetlands, floodplains estuaries, beaches, dunes, barrier islands, coral reefs and fish and wildlife and their habitat	Coastal Zone Management Act of 1972 (P.L. 92-583), 16 U.S.C. 451 et seq.	1. Coastal zone management program (§305, 306) 2. Resource Management Improvement Grants (§306A) 3. Federal Consistency Determination (§307) 4. National Estuarine Reserve Program (§315)
Department of the Interior - U.S. Fish & Wildlife Service (USFWS)	National resource trustee for: migratory birds; certain anadromous fish, endangered species, and marine mammals; and certain Federally managed water resources	Clean Water Act (P.L. 92-500), 33 U.S.C 1321 (f)(5)	Remedial Action Program (CERCLA, §104)
	Land and water conservation	Land and Water Conservation Fund Act (P.L. 88-578), 16 U.S.C. 460 I-4-460I-11	Establishment of fund to acquire land, waters, or interests in land or waters to promote outdoor recreation opportunities
	Coastal barrier islands	Coastal Barrier Resources Act of 1982 (P.L. 97-348), 16 U.S.C. 3501-3510	1. Establishment of coastal barrier resources system. 2. Coverage of undeveloped coastal barriers, including associated aquatic habitats
	Threatened and endangered species and their critical habitat	Endangered Species Act of 1973 (P.L. 93-205), 16 U.S.C. 1531-1543	Insurance that any action authorized, funded or carried out by any Federal Agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of habitat to such species (§7) (covers nonmarine species)
	Estuarine areas	Estuarine Areas Act (P.L. 90-454), 16 U.S.C. 1221 et seq.	Conservation of estuarine areas
	Fish and wildlife conservation	Fish and Wildlife Coordination Act of 1958 (P.L. 85-624), 16 U.S.C. 661-666c	Consultation when Federal agency or Federal permittee proposes to modify a body of water
		Fish and Wildlife Conservation Act of 1980 (P.L. 96-366), 16 U.S.C. 2901 et seq.	Conservation and promotion of nongame fish and wildlife and their habitats
		National Wildlife Refuge System Administration Act (P.L. 91-135), 16 U.S.C. 668dd	Resource management programs for fish and wildlife habitat
	Wetlands conservation	North American Wetlands Conservation Act (P.L. 101-233)	1. Funding for purchase of critical wetlands in the U.S., Canada and Mexico 2. Matching funds for wetlands conservation projects in North American
Other Department of the Interior (DOI)	Development of outer continental shelf, subject to environmental safeguards	Outer Continental Shelf Lands Act (P.L. 93-627), 43 U.S.C. 1331 et seq.	Responsible for removal of oil and gas platforms in Federal waters including those used as artificial reefs
Council on Environmental Quality (CEQ)	Major Federal actions significantly affecting environmental quality	National Environmental Policy Act (P.L. 91-190), 42 U.S.C. 4321 et seq.	1. Mediate interagency disputes
U.S. Army Corps of	Wetlands protection	Clean Water Act (§404)	Dredge and fill permits

FEDERAL AGENCY	SCOPE OF RESOURCE AND HABITAT MANAGEMENT RESPONSIBILITY	LEGISLATIVE AUTHORITY	POTENTIAL PROGRAMS
Engineers (USACOE)		(P.L. 92-500), 33 U.S.C. 1251 et seq.	
	Wetlands creation	Water Resources Development Act of 1976 (§150) (P.L. 94-587), 42 U.S.C. 1962d-5e	Authority to establish wetland areas as part of an authorized water resources development project
	Beach nourishment	Water Resources Development Act of 1976 (§150) (P.L. 94-587), 42 U.S.C. 1962d-5f)	Authority to utilize suitable dredged material for beach nourishment
	Avoiding obstructions to navigation	Rivers and Harbors Appropriation Act of 1899, 33 U.S.C. 401	Regulation of construction activities in an adjoining navigable water which alter the course condition, location, or capacity of such waters
	Regulation of dredged material ocean dumping	Marine Protection Research and Sanctuaries Act (§103) (P.L. 92-532), 33 U.S.C. 1401 et seq.	1. Issuance of ocean dumping permits (§103) 2. Ocean dumpsite selection (§103)
	Fish and wildlife mitigation	Water Resources Development Act of 1986 (§906) (P.L. 99-622), 33 U.S.C. 2201, 2283	Mitigation of fish and wildlife losses associated with authorized water resources projects, including the acquisition of lands or interests in lands
Food and Drug Administration (FD) and Department of Health and Human Services (DHHS)	Healthfulness of fish and shellfish marketed in interstate commerce	Federal Food, Drug and Cosmetic Act, 21 U.S.C. 301-392	1. Setting standards of quality for foods, including seafood (§401) 2. Setting action levels and tolerances for unavoidable contaminants in foods including seafood (§406)
U.S. Department of Agriculture (USDA)	Wetlands protection	Water Bank Act (P.L. 91-559), 16 U.S.C. 1301, 1311, 1501, 1503	3. Preserve, restore, and improve wetlands; conservation easements

2.5.1 Federal, Legal, and Regulatory Constraints

Exhibit 2.23 provides an extensive, but not exhaustive, catalog of the federal authorities and programs most likely to affect the implementation of a restoration action. The key elements of the federal programs are identified, including the scope of each agencies' responsibilities, legislative authority, and specific program area(s). Any individual restoration action may come within the purview of several federal agencies and programs. These programs range from broad, national programs (e.g., Marine Mammal Protection Act), to geographically limited or species-specific initiatives (e.g., Atlantic Striped Bass Conservation Act). In general, regulatory factors can be segmented by agency for which there are requirements for consultation for formal permits. While this listing is a helpful "checklist," it must be recognized that the ultimate breadth and significance of these and other regulatory factors is highly site- and resource-specific and should not be generalized or assumed.

The following are examples of the federal permits that may be required to implement a preferred restoration strategy:

- The gathering of wild marsh plants or seeds from federal lands requires a permits from the federal agency with management responsibility at the proposed collection sites;

- Subtidal bottom restoration activities involving dredging of sediments, or the capping of contaminated sediments in place, require dredge and fill permits (known as "Section 404" permits) from the U.S. Army Corps of Engineers (USACOE). A distinct permitting process, established by Section 103 of the National Marine Sanctuaries Act, applies to restoration actions that require ocean disposal;

- Restoration alternatives that entail the taking, breeding, or releasing of marine mammals are subjects to the extensive review and permitting requirements of the Marine Mammal Protection Act. In a similar fashion, the Endangered Species Act requires permits when capturing, unintentional taking, breeding, or releasing of endangered resources is involved;

- Restoration actions involving artificial reefs are subject to a USACOE permit associated with the alteration of navigable waters. Artificial reefs could also involve the U.S. Coast Guard if navigation safety issues are involved, the Minerals Management Service if an abandoned oil and gas rig is proposed, or the Maritime Administration if an obsolete U.S. merchant marine vessel is at issue;

- Restoration construction actions involving wetlands adjacent to the territorial seas or waterways and their tributaries are subject to the USACOE permitting process. Restoration alternatives using dike-type devices to control erosion are subjected to this same permitting procedure. In a similar fashion, a new fish hatchery project connected to a navigable waterway requires a USACOE permit; and

- Bioremediation in the coastal zone requires an EPA discharge permit. If the proposed restoration action is considered experimental, there could be substantial delays while the advice of other departments and the scientific community is solicited. For example, in the *Exxon Valdez* experience, it was reported that four months elapsed before the necessary permits for bioremediation were approved (Chianelli, 1992).

A second broad category of regulatory concern typically consists of some form of general consultation with other federal agencies that have statutory jurisdiction or interest over some aspect of the resource. Examples of the range of other federal resource management concerns that may apply to specific resource restoration actions follow:

- The National Estuary Program Office's program created by the Clean Water Act (i.e. Chesapeake Bay Program) has responsibility over actions which would affect environmental quality throughout an estuary;

- The Coastal Barrier Resources Act of 1982 gives DOI authority to restrict development within the Coastal Barrier Resources System (CBRS). Restoration actions involving any of the 452,834 acres in the CBRS require DOI concurrence;

- The Water Resources Development Act, Land and Water Conservation Fund Act, and the Waterbank Act authorize the USACOE, DOI, and Department of Agriculture, to acquire, reserve, restore, or establish conservation easements for wetlands. Off-site wetland restoration actions conducted under these initiatives should be consistent with ongoing local initiatives;

- Marine sanctuaries, national parks, wilderness areas, and national forests are a few examples of special natural resource management areas. Restoration actions in special management areas require the concurrence of the appropriate program office; and

- Fishery restoration actions involving management or restocking may be subject to various fisheries management programs, such as regional Fishery Management Councils, the Salmon and Steelhead Conservation and Enhancement Act, or the Atlantic Salmon Conservation Act.

Many programs with statutory authority over natural resources fall within NOAA, USDA, and DOI, the same agencies actively involved in the damage assessment and restoration planning processes. EPA also has many statutority mandates affecting natural resources. Because of this, an effective interagency review of a draft restoration plan should reflect the necessary inputs from many of the federal programs with an interest in the restoration action. However, there remains a potential range of other federal programs or initiatives with the authority to delay or complicate implementation of a restoration action inconsistent with their statutory authorities.

Exhibit 2.24 indicates whether particular programs have regulatory and management, funding, acquisition, or research authority. The following key explains how an understanding of these federal programs can be used to plan a restoration strategy:

- Regulatory and management programs typically have the authority to directly regulate or permit specific activities;

- Acquisition-type programs may exist in federal offices where parallel restoration and habitat enhancement alternatives are ongoing, synergies may exist from coordinating with these initiatives; and

- Research/monitoring programs are those primarily involved in examination or experimentation. These offices may be both sources of scientific support or have an interest in the research aspect of quasi-experimental restoration actions.

2.5.2 State and Local Legal and Regulatory Constraints

In addition to the above federal programs, restoration actions must also be consistent with an often equally extensive range of state or local regulations. At a general level, many of the state regulatory factors closely track with the above federal programs. For example, many state Departments of Fish and Wildlife follow the guidelines of the U.S. Fish and Wildlife Service. There are also a number of joint state/federal programs, such as the Chesapeake Bay Program, in which federal and state priorities and regulatory initiatives are considered fully integrated through contacts with the appropriate program office. However, there are situations in which state or local regulatory conditions diverge from those in the federal or other states. For example, some states specifically ban dispersant use for cleanup or restoration actions.

There are also a variety of state or local permitting programs. Because of the large number of permutations among the various states and hundreds of coastal counties, permitting factors related to restoration at this level are not presented in this document.

Exhibit 2.24 Focus of federal program roles potentially affecting implementation of restoration.

Resource	Legislative Program	Lead Agency	Management / Regulatory	Funding	Acquisition	Research/ Monitoring	Likely Significance to Restoration
Fish	Anadromous Fish Conservation Act	NOAA USFWS	X	X	X	X	
	Salmon & Steelhead Conservation & Enhancement Act	NOAA	X		X	X	
	Magnuson Fishery Conservation and Management Act	NOAA	X	X			
	National Fishing Conservation and Management Act	NOAA	X	X			
	Fish Restoration and Management Project Act	USFWS		X			X
	Atlantic Salmon Conservation Act of 1982 (P.L. 97-389), 16 U.S.C. 3601-3608	NOAA	X				
	Atlantic Striped Bass Conservation Act (P.L. 89-304), 16 U.S.C. 757 g	USFWS		X		X	
Shellfish	National Shellfish Sanitation Program, 16 U.S.C. 1642nt	FDA	X			X	
Mammals	Marine Mammal Protection Act	NOAA	X	X		X	X
	Fur Seal Act	NOAA	X			X	
Waterfowl and Other Birds	Migratory Bird Conservation Act	USFWS	X		X		X
Wetlands	North American Wetlands Conservation Act	USFWS		X	X		X
	Water Resources Development Act (Wetlands Creation)	USACOE	X		X		X
	Water Bank Act	USDA	X	X	X		X
Estuarine Areas	Clean Water Act (National Estuary Program)	EPA	X	X		X	
	Coastal Zone Management Act (National Estuarine Reserve Program)	NOAA	X	X	X	X	
Barrier Islands	Coastal Barriers Resources Act, 16 U.S.C. 3501-3510	USFWS	X				
Marine Sanctuaries	National Marine Sanctuaries Act	NOAA	X	X			
Surface Waters, Wetlands, and Aquatic Biota	Clean Water Act	EPA USACOE (*404)	X			X	X
Ocean Water and Marine Biota	Marine Protection, Research, and Sanctuaries Act (Title I)	EPA USACOE	X			X	
Coastal Resources	Coastal Zone Management Act	NOAA	X	X			
Water and Resources of the Outer Continental Shelf	Outer Continental Shelf Lands Act, 43 U.S.C. 1331-1356	Minerals Management Service	X	X		X	
Endangered Species and Their Critical Habitat	Endangered Species Act	USFWS NOAA	X	X		X	

Fish and Wildlife and Their Habitat	Fish and Wildlife Coordination Act	USFWS	X			X	
Safety of Commercially Marketed Fish and Shellfish Products	Food, Drug & Cosmetic Act	FDA	X				

EFFECTIVENESS AND SUCCESS CHAPTER 3

3.1 Overview

The goal of natural resource restoration should be a return to baseline conditions. This goal is achieved when the natural resource is able to maintain its normal function and services without assistance from man. Success of a restoration action is measured by comparison of a restored natural resource's ecological structure and function to the characteristics of natural resources of the same type and geographical region. Comparisons may involve baseline (i.e., pre-incident) data on the same site or control (or reference) data relative to the affected site. Natural variability of a habitat or natural resource in space and time should be incorporated into this comparison using valid statistical methods, as data are available. A good scientific design should be used, i.e., hypotheses should be stated clearly, tested with appropriate statistical design and analysis, and results quantitatively presented. The null hypothesis will normally be that oil-affected natural resources have the same structure and function as baseline conditions (i.e., as they would be without the incident having occurred). Statistical testing must be powerful enough to reject this null hypothesis and accept the alternative hypothesis, that there are differences between the oiled and non-oiled condition of natural resources, to assess both injury and the degree of recovery. Improvement of the rate of recovery induced by some restoration action is a measure of effectiveness of the action.

The degree to which functional replacement of natural systems is achieved determines the effectiveness of restoration. The ability of a habitat or population to maintain proper functioning and persist over time also needs consideration. The functions of a habitat will differ by type and location, but generally include:

- Biological diversity;
- Shellfish and finfish habitat;
- Wildlife habitat;
- Food chain functions/productivity/trophic bioaccumulation;
- Hydrology (water storage/conveyance, groundwater);
- Pollution control - sediment trapping (wetlands);

- Capacity to remove nutrients, contaminants and toxins from run-off and effluent (wetlands);

- Recreational use;

- Commercial use;

- Management areas;

- Extraction sites;

- Cultural sites;

- Education and research; and

- Aesthetics.

Clearly stated goals of restoration should be developed, and alternatives and actions evaluated relative to those goals. It is the function and not merely appearance of a habitat which the trustees must consider in monitoring of a restoration project. Also, interconnections between the habitat and surrounding natural resources must be restored (Cairns, 1991). Monitoring and comparison with naturally occurring systems should include evaluation of:

- Degree of injury to individual species and populations;

- Revegetation rates and species composition;

- Repopulation by fauna (particularly ecologically important ones);

- Redevelopment of soil profiles;

- Ecosystem services (e.g., productivity, carbon and nutrient storage);

- Patterns of succession; and

- Evidence of persistence (i.e., long term viability, unaided) of the habitat.

Westman (1991) outlines a protocol for measuring success of restoration projects:

- Define restoration goals;
- Select appropriate criteria for monitoring goal achievement;
- Identify performance standards; and
- Measure levels of achievement of these standards.

He suggests several methods for measuring achievement, both quantitative and graphical (See Westman, 1991).

While the goal of restoration ideally is to return all function and services to "normal" state, this may not be possible. Thus, the trustees need to identify and prioritize the functions and services. Recovery should be measured relative to the condition that would be achieved had the discharge not occurred. A baseline, reference, and/or control site serves as a proxy to the condition that would exist if there were no injury.

Evaluation should include success of biotic establishment and use of the area by species of concern. Success of restoration should be measured during the implementation of restoration and over the long term to measure permanence of restoration. Length of monitoring should be sufficient to determine the return of all necessary habitat functions and the ability of the habitat to maintain these functions. Statistically sound sampling strategies are a must for making these determinations.

The following sections contain an evaluation of restoration effectiveness and success based upon existing information. A review of individual documents is followed by a synthesis of the state-of-the-science for restoration actions on the habitat and/or resources. Recovery rates after various actions are quantified where possible and incorporated into the review of effectiveness. In addition, the risks and hazards associated with procedures are considered. These include the additional injury to an ecosystem caused by restoration practices, injuries to adjacent and associated natural resources as well as possible risk to project personnel.

The review below includes all oil-related literature that were available. For a few habitats, there is considerable information. However, for most habitats, little or no documentation of natural recovery or restoration after oil discharges is available. In most cases, the information is anecdotal.

Further, the distinction between response and restoration actions is not made in this literature. In fact, most information concerns follow-up to response rather than restoration in its true sense. However, natural recovery following no action and various response actions is pertinent to the analysis of restoration actions. Thus, the literature on both response and restoration efforts is thoroughly reviewed to make the most informed analysis and recommendations regarding recovery and success.

In this section, the terms response and cleanup are used if that was the context of the literature report. Response is used as a general term including all activities performed immediately following the discharge by response agencies. Response activities include cleanup, which is defined as purposeful removal of oil or acceleration of natural removal processes. The term cleanup is used when specifically describing such activities.

The approach taken was to review available information available for oil discharges, plus additional information in the much vaster restoration literature as it applies to oil discharge situations. Habitat restoration is reviewed in Section 3.2 for nine classes of habitats that have distinct approaches to restoration. Section 3.2.10 discusses monitoring of habitats in general. Specific information on each habitat that is not generally applicable is in the individual sections (3.2.1 to 3.2.9). This avoids repetition of concepts, since much of the information is generally applicable. Biological natural resources (i.e., species populations) restoration is reviewed in Sections 3.3.1 to 3.3.5.

The section on each habitat or biological natural resource is organized as follows. Case histories of oil discharges and non-oil restoration efforts are first reviewed. Second, experimental studies of oiling, recovery, and restoration effectiveness are evaluated. Finally, the information is summarized and conclusions are made in subsections entitled " Restoration and Recovery: Summary and Conclusions." This evaluation focuses on effectiveness and success. In Chapter 5, evaluation of restoration alternatives and actions is made considering technical feasibility, effectiveness and success, and cost.

It should be emphasized that the recommendations made in the following sections are intended to provide guidance as a synthesis of available information. It is not intended that this document be a cookbook for restoration. Specifics of the site, discharge, and context of the situation will need careful consideration and may drive the decisions made. What is presented here is an analysis of the effectiveness of various alternatives and actions available.

3.2 Habitat Restoration and Mitigation

3.2.1 Estuarine and Marine Wetlands

3.2.1.1 Saltmarshes

Saltmarshes are low-energy intertidal habitats that are particularly vulnerable to oil discharges. A large literature on discharge impacts, mitigation, restoration, remediation and recovery is available for these habitats. Most studies concern acute impacts following single discharges and focus on marsh vegetation, particularly *Spartina* spp. Saltmarsh fauna were monitored intensively in studies following the September 1969 West Falmouth discharge, but typically are not evaluated. The only well studied chronic oiling situation identified in this review is that of the Fawley marsh in the U.K. Case studies of oil discharge incidents in saltmarsh habitats are reviewed below in chronological order. Experimental studies of the effects of oil, oil response activities, and marsh restoration methods are discussed separately in Section 3.2.1.1.2. Restoration studies of non-oiled marshes are reviewed in Section 3.2.1.1.2.9. In all literature cited in this document, the units of measurement used by the individual authors cited are used.

3.2.1.1.1 Case Studies of Oiling in Saltmarsh Habitats

3.2.1.1.1.1 *Chryssi P. Goulandris* Discharge

In January, 1967, the tanker *Chyrssi P. Goulandris* discharged 250,000 kg of light Kuwait crude oil. The oil reached the Bentlass saltmarsh in Pembrokeshire, Wales, where it covered low marsh areas and penetrated deep into marsh sediments. Response activities involved bulldozing lower shore gravels, cutting and removal of oiled *Spartina*, and widespread use of detergent and surfactant sprays.

Cowell (1969) and Cowell et al. (1969) monitored the marsh for a period of one year following the discharge, performing frequency analyses to determine statistically significant changes in plant species composition after the discharge. Pre-incident data were available because a permanent teaching transect was located in the marsh. One month after the discharge, permanent quadrants were established in the marsh. Vegetation was surveyed at the time the quadrants were established, the following spring, and again one year later. Statistical analysis was not discussed and apparently hypothesis testing was not performed.

Direct measurements of oil were not made. Cowell (1969) and Cowell et al. (1969) reported that no oil was visible in the marsh one year after the discharge. In areas affected by surfactants in combination with oil, plant mortality was approximately twice as great as with oil alone. Five months after the discharge, some species of marsh plants were not affected while others exhibited impaired seed germination. One year after the discharge, the annual plants, *Suaeda maritima* and *Salicornia* spp., recovered to some extent, but not to pres-incident levels. *Spartina townsendii*, which was in its winter dormant stage at the time the discharge occurred, recovered completely one year after the discharge.

3.2.1.1.1.2 *Torrey Canyon* Discharge

The tanker *Torrey Canyon* discharged 60,000 metric tons of Kuwait crude oil in March, 1967. Approximately 20,000 tons of weathered oil reached saltmarsh habitats in the Hayle and Gannell estuaries of Cornwall 7-8 days after the discharge incident. Another ~20,000 tons of oil came ashore immediately along the Brittany coast of France. In Cornwall, a small portion of the Hayle marsh was cleaned using unspecified methods (Cowell, 1969). Surfactants were not used. In Brittany, cleanup involved mechanical removal of oiled marsh soil: in the St. Anne marsh, 1 ha of the top 15-20 cm of soil was removed; in the Perros-Guirec marsh, 1 ha was covered with 2-3 m of oiled sand and domestic refuse (Stebbings, 1970).

The oil, which was weathered by the time it came ashore, was distributed in discontinuous patches in Cornwall. Some pre-incident data were available. Three months after the discharge, no broad-scale effects on marsh vegetation were visible except in small, localized patches (Cowell, 1969). Statistical analysis was not discussed and apparently hypothesis testing was not performed.

The Brittany marshes suffered considerably more injury than those in Cornwall. Stebbings (1970) visited two marshes in Brittany immediately after the discharge and again sixteen months later. Statistical analysis was not performed, and apparently hypothesis testing was not done. Stebbing's (1970) qualitative observations indicated that fourteen days after the discharge, both the St. Anne marsh and the Peros-Guirec marsh were coated with a layer of thick, heavy oil. Sixteen months later, vegetation in the St. Anne marsh appeared normal, exhibiting luxuriant growth. However, oil was still visible over the marsh surface and penetrated 3 cm into marsh litter and soil, producing reducing conditions beneath this depth. The species composition of the lower marsh vegetation changed to a monoculture of *Triglochis maritima*. After sixteen months, vegetation in the Perros-Guirec *Spartina* marsh appeared healthy, with most plants in flower. Thick oil was present to a depth of 15-20 cm in the sediments and living plant roots were present only in new sand above that level. The species composition of the lower marsh changed. Stebbings (1970) considered the shifts in plant species dominance short-term and noted that some species (i.e., *Juncus gerardii*, *Triglochis maritima*, *Halimione portulecoides* and *Paccinellia maritima*) were particularly successful in withstanding oiling. The time required for full recovery of the vegetation was not estimated.

3.2.1.1.1.3 The West Falmouth Discharge

A small discharge of No. 2 fuel oil at West Falmouth, Massachusetts in September 1969 contaminated contiguous saltmarshes at Wild Harbor with up to 6,000 mg oil g^{-1} sediment (Krebs and Burns, 1977). Mass mortalities of invertebrates occurred immediately. Emulsifiers were used to disperse oil in waters south of Wild Harbor, but their use was discontinued after a few days because of shellfish toxicity (Sanders et al., 1980). No other response activities were reported.

The persistence of petroleum compounds from the discharge in marsh sediments and biota was monitored for periods ranging from 5 (Michael et al., 1975; Sanders et al., 1980) to 20 (Teal et al., 1992) years. In contrast to the majority of post-discharge monitoring efforts which concentrate on saltmarsh vegetation, intensive long-term studies of benthic organisms were performed at Wild Harbor marsh (Michael et al., 1975; Krebs and Burns, 1977; Sanders et al., 1980).

Sanders et al.'s (1980) 5-year monitoring study along an onshore/offshore gradient was designed to statistically evaluate whether persistent, deleterious effects occurred in benthic organisms as a result of the oil discharge. Hydrocarbons were measured in sediments and biota at 6 West Falmouth stations characterized by varying degrees of oiling and at a reference station in unoiled Sippewissett marsh. Oil concentrations remained high in intertidal and subtidal peat and mud of the Wild Harbor River for the duration of the five year study. Changes in density, number of species, and species diversity of benthic organisms were most pronounced in areas heavily oiled. After five years, the fauna had only partly recovered.

Petroleum hydrocarbons attributable to the discharged oil persisted in sediments in some parts of the marsh for 20 years (Teal et al., 1992). The chemical composition of petroleum hydrocarbons changed over time, as did their degree of penetration into marsh sediments. Alkanes disappeared after about four years, while heavy aromatics and napthenes persisted for at least 8 years. In 1971, two years after the discharge, oil penetrated the sediment to a depth of 70 cm; by 1975, 7 years after the discharge, no oil was observed below 20 cm. All organisms analyzed exhibited initial high contamination. *Fundulus* was nearly free of contamination after one year, but the marsh crab *Uca pugnax* remained heavily contaminated for at least four years (Burns and Teal, 1979). Teal et al. (1992) sampled sediments from five of the original stations and two of the original reference sites in August 1989, 20 years after the discharge. There was no evidence of fuel oil at three of the stations. However, one subtidal mud core contained traces of biodegraded fuel oil at 10-15 cm, and one marsh core contained $10^{-6}g.g^{-1}$ dry weight of weathered and biodegraded fuel oil aromatic hydrocarbons and cycloalkanes at 5-10 cm with lesser concentrations at 0-5 and 10-15 cm. Thus, some oil attributable to the discharge persisted in relatively high concentrations in sediments in the most heavily oiled area of the marsh 20 years after the discharge. Overall, less than 1% of the marsh remained significantly contaminated. Levels of microsomal cytochrome P4501A, which is induced by hydrocarbons, were elevated in *Fundulus* collected from Woods Hole versus the reference sites in 1989. However, between-site differences were not large and were only marginally significant, indicating that present-day fish in the area are coming into contact with small amounts of oil from sediments contaminated 20 years ago.

Teal et al. (1992) reported that the marsh is now visually no different from other healthy New England saltmarshes, provided that the oiled area remains undisturbed. Any severe disruption of marsh sediments in the area still contaminated could release sufficient oil to have observable local effects, the magnitude of which would depend on the rapidity with which the released oil was dispersed. Teal et al. (1992) noted that an animal burrowing into the still contaminated sediment would be exposed to oil concentrations that caused significant biological effects in the past. Whether burrowing animals now avoid the area or are still burrowing there and being killed as occurred during the year following the discharge is unknown.

3.2.1.1.1.4 Buzzards Bay Discharge

In October 1974, the oil barge *Bouchard 65* discharged an undetermined amount of No. 2 fuel oil off the west entrance of the Cape Cod Canal in Buzzards Bay, Massachusetts. Over the following two weeks, oil was found in saltmarsh habitats located about 5 km from the discharge site. Massive mortalities of invertebrates (i.e., seaworms, gastropods and decapods) were observed immediately following the oiling (Hampson and Moul, 1978).

A transect was established in the Windsor Cove marsh that was monitored immediately after the discharge and again three years later. Because pre-incident information was not available, a nearby unoiled marsh was used as a reference site. Statistical analysis was not discussed, and apparently hypothesis testing was not done. Yellowing of *Spartina alterniflora* leaves was observed immediately following the discharge. After three years, *S. alterniflora* in lower marsh areas did not reestablish by either reseeding or rhizome growth. Marsh sediments showed a correspondingly high concentration of oil in the peat substrate, and erosion rates over the three year period were 24 times greater than those measured in the reference marsh. *Salicornia virginica* recovered to some degree in the higher marsh areas. The marsh mussel *Modiolus demissus* recovered from the discharge. Unrestricted by *Spartina* root systems, after three years mussel numerical abundances were higher in the oiled marsh than in the reference marsh (Hampson and Moul, 1978). The time required for full recovery of the oiled marsh was not estimated.

3.2.1.1.1.5 Hackensack Meadowlands Discharge

In May 1976, a ruptured fuel tank discharged two million gallons of No. 6 fuel oil into the Hackensack River, New Jersey. The flood tide carried the oil about 4 km upriver into the Kingsland Creek-Sawmill Creek area of the Hackensack meadowlands. A combination of winds and currents deposited most of the oil in back marsh and mudflat areas along the west bank of the river. Response activities consisted of cutting and removing oiled vegetation 5-15 cm above the soil surface in the most heavily contaminated areas along the river banks. No cutting was done in inaccessible soft mud environments (Mattson et al., 1977; Dibner, 1978). Approximately 8,000 feet of riverbank was cut, equivalent to ~11% of the oiled shore area (Mattson et al., 1977).

The marsh was monitored four times during the year following the discharge. Nine marsh and two mudflat stations were sampled. Sites included areas oiled and cut and areas oiled and not cut. In the marsh sites, vegetative cover, stem density, stem height, invertebrate fauna, and sediment were sampled with replication. Erosional data were collected at all vegetated stations (Dibner, 1978). Although basic descriptive statistics were calculated, statistical analyses involving hypothesis testing were not performed.

After one year, mortality was highest in heavily oiled *Spartina* plants that were not washed clean by the tide or cut. Cutting heavily oiled plants soon after contamination was beneficial and reduced long-term damage to the plants, despite trampling. Trampling was detrimental. Highly trampled banks became more susceptible to erosion, with severe erosion restricted to the cut regions (Dibner, 1978). The time required for full recovery was not estimated.

3.2.1.1.1.6 *Amoco Cadiz* Discharge

In March 1978, the supertanker *Amoco Cadiz* broke up off the coast of Brittany, discharging 223,000 tons of light crude oil (Bellier and Massart, 1979). A layer of oil up to 30 cm deep covered the Ile Grande saltmarsh on the north coast of Brittany over a two week period. Because of the heavy oiling, the inner part of the marsh was considered to be beyond natural recovery. A massive response effort involved the use of heavy machinery to remove oiled vegetation and a large amount of the oiled surface sediment from both banks of the main marsh channel, in some areas to a depth of 30-50 cm. Many of the primary and secondary channels draining into the main marsh were excavated, widened or deepened in an effort to drain oil trapped on the upper marsh (Vandermeulen et al., 1981; Long and Vandermeulen, 1983; Baca et al., 1987).

Removal of marsh sediment during cleanup activities altered the geomorphology of the marsh, resulting in a marked increase in the marsh cross-sectional area, in its tidal prism, and in tidal current velocities through the marsh. Two years after cleanup, the marsh's normal net accretion rate of 28-90 cm y^{-1} had shifted to a net erosion rate of 6.5-17 m y^{-1}. Increased tidal current velocities eroded exposed marsh surfaces and undercut secondary and tertiary tidal channels. Residual oil, left behind during the cleanup, remained trapped under sandbars (Vandermeulen et al., 1981). Four years after the discharge, the marsh remained in a net erosional state (Long and Vandermeulen, 1983), but natural recovery began through invasion of annual plants and rhizome spreading of perennials. Opportunistic species had increased (Baca et al. 1987).

A large-scale transplantation effort began one year after the discharge, following preliminary experiments to compare types of transplants, fertilizer materials, and planting seasons and assess the feasibility of field nursery production of marsh plants. Planting continued over a 3 year period. Eventually, 12,000 field-dug and nursery grown plants were placed along creek banks and other disturbed areas. Statistical analyses involving hypothesis testing were not performed (Broome et al., 1988). Plug-type transplants (i.e., roots with a core of substrate) of *Puccinellia maritima* exhibited superior survival and growth rates compared to sprig transplants (i.e., roots without substrate), although sprig transplants grew well and survived. Fertilization with slow-release nitrogen and phosphorus was necessary for good transplant growth on disturbed sites (Broome et al., 1988; Seneca and Broome, 1992).

Baca et al. (1987) surveyed the Ile Grand marsh in the fall and spring of 1985 and 1986, 7 and 8 years after the discharge. They compared marsh sites oiled but not cleaned, sites heavily oiled and cleaned, and control sites neither oiled nor cleaned. Surveys were quantitative, with analysis of variance performed to determine statistically significant differences among sites. After 7-8 years, there were little or no significant differences in species occurrence and coverage between and among sites. The Cantel marsh, which was oiled but not cleaned, was restored within five years of the discharge. The Ile Grande marsh, which was oiled and cleaned, was restored within 8 years of the discharge. Baca et al. (1987) concluded that response and cleanup activities delayed recovery of the Ile Grande marsh by two to three years.

3.2.1.1.1.7 Barge *STC-101* Discharge

In February 1976, the barge *STC-101* discharged 250,000 gallons of No. 6 fuel oil into lower Chesapeake Bay. Most of the oil was carried across the Bay to its eastern shore in Northampton County, Virginia, where it was stranded intertidally on beaches and fringing marshes. Cleanup of the marshes began immediately, and involved cutting and removing the standing dead stems of marsh grass, taking care not to disturb the marsh peat (Hershner and Moore, 1977).

Marsh plants and invertebrates were surveyed quantitatively along transects for one growing season after the discharge. Marsh grass production and growth were measured. Because prespill data were not available, nearby reference sites were monitored for comparison. Basic descriptive statistics were calculated, but statistical analyses involving hypothesis testing were not performed. On the basis of population densities, mussels and oysters suffered no short-term effects from the oiling and snails had recovered ~8 months after the discharge. *Spartina alterniflora* exhibited a short-term increase in production and a greater rate of flowering in oiled areas (Hershner and Moore, 1977).

3.2.1.1.1.8 *Lang Fonn* Discharge

In December 1978, the Norwegian tanker *Lang Fonn* accidentally discharged 360-700 barrels of No. 6 fuel oil into the Potomac River at Piney Point, Maryland. Winds and flood tides pushed the oil along a sand spit into Piney Point Creek, where up to 600 barrels collected in a small cove. Response activities involved pumping the oil from the cove. Weather delayed this phase of the response by several weeks, during which the low marsh fringing the cove was heavily oiled. Oiled vegetation was cut and the debris raked from the marsh surface to remove contamination. Sorbents were used to remove pockets of surface oil from heavily oiled areas (Krebs and Tanner, 1981).

Krebs and Tanner (1981) performed experimental studies in the oiled marsh and an unoiled control marsh to evaluate the restoration potential of sediment stripping and replanting with propagated *Spartina*. The experimental design consisted of stratified random sampling of 12 experimental plots over two growing seasons. Experimental treatments consisted of sediment stripping and backfilling with and without subsequent replanting in oiled and unoiled areas. *Spartina* stem and shoot density, aboveground biomass and seed head production were measured monthly or bi-monthly. Snail densities were measured monthly. Mussel densities were measured only in the spring. The relative abundances of major meiofauna taxa were measured bi-monthly in the second year of the study. Sediment hydrocarbons were measured twice in the first year and once in the second year of the study. Sediment stripping had no effect on any measured *Spartina* parameters. *Spartina* transplants grew at similar rates in both the oiled and unoiled plots. By the end of the first year, heights, densities, and aboveground biomasses of transplants grown in oiled and stripped plots did not differ significantly from those grown in unoiled control plots. Backfilling did not affect growth in the first year. By the end of the second growing season, *Spartina* densities decreased along the lower areas of oiled plots, apparently in a delayed response to oiling. Numbers of benthic invertebrates were reduced after the oiling and cleanup. Snails were physically removed by sediment stripping, and populations began to recover only after a recruitment event which occurred two years after the discharge.

3.2.1.1.1.9 Houston Ship Channel Discharge

The collision of an oil barge and a tugboat discharged 42,000 gallons of No. 6 fuel oil into the Houston ship channel in October, 1977. Much of the oil washed onto fringing marshes of *Spartina alterniflora* adjacent to the ship channel. The plants were completely covered by oil. Some of the discharged oil was carried toward the Gulf of Mexico by tidal currents within the ship channel, along the north jetty, through a boat cut, and washed onto several ha of *Spartina* marsh located east of the jetty. Response activities involved use of 3M, a synthetic sorbent agent, to remove oil from marsh areas. Oiled marsh grass was cut and removed by raking and shoveling (Webb et al., 1981).

The marsh vegetation was monitored for one growing season after the discharge. Live and dead stem density, stem height, and both aboveground and belowground biomass were measured in oiled and adjacent unoiled reference sites. Basic descriptive statistics were not calculated, and statistical analyses involving hypothesis testing were not performed. By the following spring, *Spartina* growth from surviving roots in oiled sites was normal. Plants in areas heavily oiled were similar in height and appearance to those in unoiled areas. Seed production in August and September was normal. Plants growing in lightly oiled areas not cleaned appeared normal. Webb et al. (1981) concluded that complete recovery of marsh grass was achieved in one growing season.

3.2.1.1.1.10 *Esso Bayway* Discharge

In January 1979, the oil tanker *Esso Bayway* accidentally discharged approximately 6,000 barrels of light Arabian crude oil into the Neches River above Port Neches, Texas. Much of the oil was concentrated in Block Bayou on the south side of the Neches River and in two canals on the north side of the river. Oil distribution in the bayou and adjacent marshes was uneven. Neff et al. (1987) estimated that <10% of the total marsh area was affected. Response activities consisted of low pressure flushing and sorption of oil (McCauley et al., 1981).

Commercially important penaeid shrimp and sediment hydrocarbons were monitored for 11 months following the discharge. Twelve sampling stations were visited monthly for 9 months. Eight of the stations were affected to varying degrees by oil. Four of the stations were unoiled reference sites. Descriptive statistics were calculated only for hydrocarbon concentrations. Statistical analyses involving hypothesis testing were not performed. After elevben months, oiled stations had the same species diversity as unoiled sites. Penaeid shrimp were absent from marsh waters during the first six months of the study, presumably due to an extended period of high rainfall rather than to effects of oiling. The shrimp returned to the marsh in November when salinity increased. Shrimp collected at oiled and unoiled sites exhibited the same physiological and morphological "condition". Sediments from both oiled and unoiled sites contained significant amounts of petroleum hydrocarbons, reflecting the large amount of refinery activity and natural oil contamination in the area (Neff et al., 1987).

3.2.1.1.1.11 Cape Fear Discharge

In May 1976, heavy fuel oil from an undetermined source discharged into the Cape Fear River, North Carolina. Thirty miles of high marsh shoreline dominated by *Spartina, Scirpis* and *Juncus* were covered by water-insoluble, hydrophobic oil that adhered to marsh plant surfaces but not to beaches or mud flats. Baca et al. (1983) calculated the amount of oil washed ashore from the amount of marsh grass surface area. An aerial survey was performed to locate sites of major, moderate, and low impact. Ground surveys were undertaken to measure the oiled area and identify the affected vegetation. The surface area submerged at high tide and therefore subject to oiling, was determined separately for *Spartina, Scirpis,* and *Juncus* using leaf geometry. It was estimated that 175,000 gallons of oil were on shore one week after the discharge. Five months after the discharge, oil was not present in lightly oiled areas and these areas had recovered. Total plants/m^2 were reduced in heavily oiled areas and oil remained on the marsh surface and in the substrate. Basic descriptive statistics were not calculated and statistical analyses involving hypothesis testing were not performed. The time to recovery was not estimated.

3.2.1.1.1.12 Galveston Bay Pipeline Discharge

A ruptured underwater transfer pipeline released 6,720 gallons of light crude oil into Dickinson Bayou, Texas, in January 1984. Saltmarsh shorelines on both sides of the bayou were oiled to varying degrees. High tides carried the oil onto marsh surfaces and up marsh vegetation to a height of 20-30 cm. An attempt was made to clean the marshes by low pressure flushing, but this effort was abandoned due to poor weather conditions and technical difficulties. Some pockets of oil in the marshes were cleaned with sorbent sheets, but this effort was minimal because of the soft substrate. Consequently, the marshes remained largely uncleaned (Alexander and Webb, 1987).

Heavily oiled, moderately oiled, and unoiled control sites were monitored for 32 months following the discharge. Basic descriptive statistics were calculated and analysis of variance was performed to determine between-treatment differences. Growth of *Spartina alterniflora* was measured 4-5 months, 7-8 months, 16-18 months, and 32 months after the discharge. Four to five months after the discharge, oil was still visible at all oiled sites. Live stem density was lower at heavily oiled sites and there was no shoreline erosion. Seven to eight months after the discharge, no oil was visible at lightly and moderately oiled sites. Some erosion had occurred and low lived plant densities were associated with the presence of oil in marsh sediments. Sixteen months after the discharge, oil was still visible at heavily oiled sites and further erosion had occurred. Heavily oiled sites had lower plant densities. Seventeen to eighteen months after the discharge, bare areas in heavily oiled sites had more oil than vegetated areas. Thirty-two months after the discharge, oil was still present at the heavily oiled sites, and considerable shoreline erosion had occurred. Plants at all sites appeared to be normal. No erosion had occurred at lightly and moderately oiled sites. Alexander and Webb (1987) concluded that oil concentrations of less than 5 mg g^{-1} did not influence *Spartina* growth in the Dickinson Bayou marsh. The time to complete recovery from heavy oiling was not estimated.

3.2.1.1.1.13 Bay Vacherie Pipeline Discharge

A pipeline break in Nairn, Louisiana released approximately 300 barrels of crude oil into a south Louisiana brackish marsh in April 1985. A total of 57 acres of marsh was affected (Fischel et al., 1989). Booms were placed around the point of rupture to contain the oil. The vegetation and sediment surface were cleaned by low-pressure flushing with ambient estuarine water, and the oily water was pumped to trucks for disposal. Oil saturated soil and plant materials were not removed from the marsh (Mendelssohn et al., 1993). Vegetation was surveyed by a combination of remote sensing and direct survey techniques three months and one and one-half years after the discharge. Benthic organisms were monitored directly (Fischel et al., 1989). Basic descriptive statistics were not calculated, and statistical analyses involving hypothesis testing were not performed.

The oiled marsh was already highly affected by human activity at the time of the discharge. Portions of the marsh were diked and used heavily by hunters, trappers, and fishermen. One and one-half years after the discharge, areal coverage of vegetation increased by 3.2 acres, and areas of injured vegetation decreased. Portions of the marsh that previously were enclosed water bodies became open water. *Spartina patens* recovered better than *Spartina alterniflora* overall, but *S. alterniflora* recovered at some sites. Vegetation loss was greatest in those areas of the marsh affected by a combination of waterlogging, oil contamination, and marsh buggy activity. Mendelssohn et al. (1993) noted that marsh buggies used in the cleanup caused some localized plant mortality due to trampling. Fischel et al. (1989) concluded that, because of human activities, the erosional processes which were occurring at the time of the discharge would continue and large-scale recovery was not likely to occur. Mendelssohn et al. (1993) reported that marsh vegetation recovered completely four years after the discharge, with no differences in *Spartina* cover between oiled and reference sites. Remote sensing data confirmed that long-term land loss rates were not affected by the discharge.

3.2.1.1.1.14 Fidalgo Bay Discharge

In late February 1991, 30,000 gallons of Prudhoe Bay crude oil were discharged into Fidalgo Bay when a pump failed during offloading at the Texaco Refinery near Anacortes, Washington. Containment of the discharge by booms along the south shoreline of the bay resulted in heavy oiling of a portion of the south marsh. Response to the discharge emphasized minimizing access to the marsh by cleanup workers and involved comparison of several low impact techniques to remove oil from the marsh. Monitoring was undertaken over a 16 month period to track marsh recovery and document the effectiveness of various response techniques. Four transects were established representing areas affected by the discharge in different ways: an unoiled control area; a lightly oiled, trampled area; and two heavily oiled areas protected from trampling in which access was gained by boardwalks. One of the latter areas was vacuumed to remove oil from the marsh surface. The other area was flushed under low pressure and then vacuumed. Measurements included percent cover of live vegetation; below ground plant biomass, and petroleum hydrocarbon concentrations in surface sediments and sediment cores (Hoff et al., 1993).

Vegetative cover differed among the treatments over time. The dominant vegetation, *Salicornia*, budded normally in the control area. Cover was 100% by June. In the oiled areas, budding occurred later in the season and plants grew more slowly, but approached 100% cover by September. The oiled transect that was flushed and vacuumed closely resembled the control transect by July. In the second growing season, among-treatment surface vegetation differences were small. Larger differences persisted in below ground biomass, however. Oil did not penetrate the sediments deeply and most oil was located within the top 2 cm. Significant weathering occurred with most alkanes gone after one year, but PAHs still present. Hoff et al. (1993) noted that occurrence of the discharge during the vegetation's dormant season probably enhanced recovery and that the trampled area exhibited the most severe impact. Low-pressure flushing followed by vacuuming was the optimum cleaning method and did not injure vegetation or marsh sediment. No estimate of time to full recovery was made.

3.2.1.1.1.15 Chronic Oiling: Fawley Marsh, Southhampton Water, U.K.

The ESSO petrochemical refinery at Fawley, Southhampton Water, U.K. discharged oily effluents into the creek system of a *Spartina anglica*-dominated marsh from 1953 until a program of effluent quality improvement was begun in 1971. Chronic oiling from the refinery effluent coated marsh plants with a thin film of oil. By 1970 an area 1000m by 600 m was completely denuded of vegetation. Except for improvements in effluent quality, no cleanup, per se, was undertaken (Dicks, 1977; Dicks and Iball, 1981; Dicks and Hartley, 1982).

The marsh vegetation was monitored for 10 years, beginning in 1969, the year before effluent improvement began. Transects were established and monitored in 1969 and 1971 to assess injury, then monitored twice yearly from 1972-1981. Qualitative observations were reported. Basic descriptive statistics were not calculated and statistical analyses involving hypothesis testing were not performed. Extensive recovery of the Fawley marsh occurred over 10 years. Several annual and perennial species recolonized due to their ability to seed rapidly. However, the original *Spartina anglica* marsh recovered more slowly. Transplanting of *Spartina* from adjacent healthy marsh areas was begun in 1975 to aid recolonization. After 10 years, affected areas located furthest from the effluent had apparently recovered, but exhibited shifts in species composition of plants and infaunal animals. No estimate of time to full recovery was made.

3.2.1.1.2 Experimental Studies of Oiling Saltmarshes

A number of controlled experimental studies concerning effects of oil on saltmarsh plant growth rates, effects of response and cleanup methods, weathering of oil, season, number of oilings, microbiological responses to oiling, and marsh establishment methods have been published. These topics are reviewed separately below.

3.2.1.1.2.1 Oil Effects on Saltmarsh Plant Physiological Rates

Stimulation of plant growth was observed following oil discharges (e.g., Hershner and Moore, 1977). Baker (1971a) performed an experimental evaluation of marsh plant growth following treatment with 4 L m^{-2} and 8 L m^{-2} Kuwait oil precipitated atmospheric residue. Qualitatively, oiled plants were a darker green color than unoiled plants. Shoot lengths of *Festuca rubra* and dry weight of *Puccinellia* sp. increased after oiling. Baker (1971a) discussed a number of possible mechanisms for the observed increases in growth, including nutrient input from oil-killed organisms, nutrient content of the oil, growth-regulating compounds in oil, and increased nitrogen fixation following oiling. However, no conclusions regarding mechanisms were made.

Smith et al. (1981) measured the rate of CO_2 fixation of saltmarsh vegetation using portable light/dark chambers to evaluate physiological stress in marsh plots that were experimentally oiled with South Louisiana crude oil. Doses of 0.2 L m^{-2} and 8 L m^{-2} were applied to replicated 6 m^2 enclosed plots. CO_2 fixation was measured 7 and 14 days after oiling. Statistical analysis was performed to determine between treatment differences. Both oil doses decreased rates of CO_2 fixation by 63-81%. Longer term monitoring was not performed to follow recovery.

Alexander and Webb (1983) tested the effects of 4 different oil types on the growth and decomposition of *Spartina alterniflora* in a Galveston Bay saltmarsh. The oils tested were Arabian crude oil, Libyan crude oil, No. 6 fuel oil, and No. 2 fuel oil. Four treatments of each oil type were applied to 1 m^2 plots in the marsh: 1 liter applied to marsh sediment; one and one-half liters applied to sediments and the lower portions of plants; one and one-half liters applied to sediments and entire plants; and two liters applied to entire plants. Unoiled plots served as control treatments. Within a week of oiling, nylon bags containing cut *Spartina* stems were placed in the center of all unoiled and one and one-half liter treatment plots to monitor decomposition. Analysis of variance was performed to determine between-treatment differences.

The results of the Alexander and Webb (1983) study are as follows. All oils caused *Spartina* mortality within three weeks. The degree of mortality varied with oil type and extent to which oil covered the plants. No. 2 fuel oil caused the highest mortality in cases where oil was applied to the entire plant surface. After five months, plant growth in the plots treated with No. 2 fuel oil was significantly less than that in unoiled control plots, but initial recovery of the No. 2 fuel oil plots began. The live aboveground biomass of plants treated with the other three oils were the same as the controls five months after oiling. Plots clipped three weeks after oil application were recolonized after five months by the growth of new stems and seedlings, but Arabian crude oil and No. 2 fuel oil significantly reduced the emergence of new stems while increasing germination. Decomposition was not affected by any oil treatment during eight months after oiling. Time to recovery was not estimated (Alexander and Webb, 1983).

Ferrell et al. (1984) performed an experimental greenhouse study of responses to oil by two *Spartina* species. Effects on growth of a number of treatments, including weathering of oil, substrate penetration of oil, coating of plant aerial tissue with oil, continuous presence of the oil layer, duration of exposure to oil, and substratum type were evaluated in factorial design and random block experiments. Sixty days after oiling, no significant differences in *S. alterniflora* growth were observed between plants treated with weathered and unweathered Venezuelan crude oil. Application of oil to aerial tissue resulted in increased mortality accompanied by decreased stem density, aerial dry weight, and regrowth. Application of oil to shoots resulted in decreased production of new shoots. Application of oil to the water layer covering the substrate surface did not reduce aerial dry weight but increased mortality and reduced dry stem density. Regrowth was completely inhibited. Only when oil was applied directly to the substrate was there a statistical difference in growth. In *S. cynosaroides*, application of oil to new shoots had no effect on stem density, aerial dry weight or regrowth density. Application to the substrate produced significant negative effects, including increased mortality, decreased dry stem density, decreased aerial dry weight, and decreased growth. Shoot production was reduced and root masses were smaller than in unoiled treatments.

Ferrell et al. (1984) concluded that the way in which oil comes into contact with marsh plant tissue or substrate is more important than weathering prior to exposure. Oil applied to the water layer did not affect existing plants, but completely inhibited growth. Oil applied to the substrate exhibited a significant effect on the plants, but had less effect on plants grown in marsh sediments (i.e., peat) than those grown in sand, presumably because the fine textured marsh sediments reduced oil penetration.

Webb and Alexander (1985) examined the effects of 4 types of oil on *Spartina alterniflora* in a Galveston Bay, Texas saltmarsh: Arabian crude oil, Libyan crude oil, No. 6 fuel oil, and No. 2 fuel oil. Experimental treatments of each oil consisted of one liter applied to sediments, one and one-half liters applied to sediments and the lower 30 cm of plants, two liters applied to sediments and entire plants, and a control treatment in which no oil was applied. Oil was applied in autumn and plant growth was evaluated after five months, one year, and two years. Analysis of variance was performed to determine between-treatment differences. All oils killed the aboveground portions of plants when applied to the entire plant surface. Partial oiling was detrimental only with No. 2 fuel oil. All types of oil applied to sediments had no effect on *Spartina*. Five months after treatment, new root and rhizome growth occurred in plants treated with Arabian crude oil, Libyan crude oil, and No. 6 fuel oil. Significantly less growth occurred in plants treated with No. 2 fuel oil. One year after oil treatment, plants treated with Arabian crude oil, Libyan crude oil, and No. 6 fuel oil had recovered completely. Plants treated with No. 2 fuel oil exhibited significantly less growth than controls. Two years after oil treatment, plants treated with No. 2 fuel oil had recovered completely. The observed slow recovery of plants after treatment with No. 2 fuel oil was attributed to initial belowground mortality rather than to long-term oil retention in the sediments.

3.2.1.1.2.2 Seasonal Effects of Oiling

Alexander and Webb (1985) evaluated seasonal responses of *Spartina alterniflora* to oil in experimental plots in a Texas saltmarsh. Four types of oil, Arabian crude oil, Libyan crude oil, No. 6 fuel oil, and No. 2 fuel oil, were applied to plants during November or May. Experimental treatments of each oil consisted of one liter applied to sediments, one and one-half liters applied to sediments and the lower 30 cm of plants, two liters applied to sediments and entire plants, and a control treatment in which no oil was applied. Live plant biomass and residual oil were measured periodically following treatment. Analysis of variance was performed to determine between-treatment differences.

No influence of season was observed by Alexander and Webb (1985) when any of the oil types was applied to sediments and lower plant parts. Reduction in live plant tissue occurred only with No. 2 fuel oil. Season influenced plant response when oil was applied to whole plants. Live plant biomass was reduced for a longer period when oil was applied in May. The greatest decrease occurred with No. 2 fuel oil. Alexander and Webb (1985) concluded that: season need not be considered for Gulf Coast saltmarshes when only sediments or parts of *Spartina* are oiled, complete oiling of *S. alterniflora* during seasons of increased growth caused longer-term reduction in live plant biomass than complete oiling during seasons of dormancy, and cleanup is warranted for discharges of No. 2 fuel oil and for discharges of all types of oil resulting in complete plant coverage during the growing season.

Baker (1971b; 1971c) performed a series of experiments in which Kuwait crude oil was sprayed on a Welsh saltmarsh at different times of year. The field experiments were supplemented with greenhouse studies. Eighteen liters of Kuwait crude oil was applied to each of three 2m x 18m transects, a dose equivalent to light oiling. Basic descriptive statistics were calculated and analysis of variance was performed to determine between-treatment differences. Most perennial marsh plant species suffered no long-term injury. The annual species *Suaeda maritima* and *Salicornia* sp., which do not possess underground roots, were injured by summer spraying. All plants exhibited a marked reduction in flower production if oiling occurred while flower buds were developing. Winter oiling of seeds reduced germination of some species in the spring. Overall, more adverse effects occurred when oil was applied during warm weather. However, recovery was rapid, regardless of the season when oil was applied. Plants oiled in May recovered by September, plants oiled in August recovered by October, and plants oiled in November recovered by the following spring.

3.2.1.1.2.3 Effects of Successive Oilings

Baker (1973) evaluated the effects of successive oilings on the recovery of vegetation in a Welsh saltmarsh. The experimental design was a random block of five 2m x 5m plots located at each of three elevations in the marsh. Treatments included 2, 4, 8, and 12 successive monthly sprayings with 4.5 liters of fresh Kuwait crude oil. Vegetative cover was recorded between oilings and at intervals over five years. Basic descriptive statistics were calculated and analysis of variance was performed to determine between-treatment differences.

Marsh plant responses to successive oilings were species-specific (Baker, 1973). For example, *Spartina anglica* recovered well by recolonizing from adjacent unoiled areas. In contrast, *Puccinellia maritima* showed little recovery on plots oiled 8 and 12 times. *Juncus maritimus* was reduced in all oiled plots located in upper marsh areas. Overall, marsh vegetation exhibited good recovery from up to 4 successive oilings, but underwent considerable changes in species composition following 8 to 12 successive oilings. In the latter cases, the changes persisted for at least five years following oiling.

3.2.1.1.2.4 Effects of Weathered Oil

Bender et al. (1977; 1981) performed experiments to determine the effects of fresh and artificially weathered south Louisiana crude oil on physically isolated plots in a York River, Virginia saltmarsh. All trophic levels were considered. Five 810 m^2 contained experimental marsh units were constructed. Four of the units were dosed with oil. One unit served as an unoiled control treatment. Measurements were made of phytoplankton standing stock, phytoplankton production, vascular plant standing stock and dry weight, snail abundance, and infaunal invertebrate abundance over 43 weeks. Analysis of variance was performed to determine between treatment differences. Both weathered and unweathered oil had similar effects on *Spartina alterniflora*: standing stocks were lower than those in the unoiled control treatment. Following initial declines after oiling, snail abundances in all oiled areas were the same as those in the control area after 43 weeks. Effects on infaunal invertebrates were less clear because seasonal changes could not be separated clearly from the toxic effects of oil.

Additional support for the contention that weathered oil is at least as toxic to plants as fresh oil comes from recent work by R. Thom (U. Washington and Battelle NW Labs). Weathered oil was found to be more toxic to kelp than fresh oil (Helton, 1993).

3.2.1.1.2.5 Effects of Response and Cleanup Methods

The advantages and disadvantages of response methods following oil discharges were reviewed by Westree (1977) and Booth et al. (1991). A number of methods were evaluated experimentally in detail to assess their effects on saltmarsh vegetation. They are discussed separately below. All of the studies cited below involved experimental oiling of marsh vegetation, statistical experimental design, and statistical analysis.

Sorbents. Sorbents reduce the possibility of recontamination by removing oil. Westree (1977) noted that sorbent materials must be recovered and removed from affected marsh areas, with the associated possibility of physical disturbance. Westree (1977) recommended that sorbents be deployed and retrieved from boats in order to avoid disturbance. Kiesling et al. (1988) reported that sorbents removed only some, not all, oil from marsh habitats.

Flushing. Low pressure flushing moves oil out of marsh areas without injury to plants or substrate and can be widely applied to all marsh and oil types (Westree, 1977). Kiesling et al. (1988) reported that low pressure flushing was effective in removing oil from marsh sediment surfaces if performed before oil penetrated the sediments. In Kiesling et al.'s (1988) experiments, No. 2 fuel oil was reduced to background levels by flushing and by flushing in combination with dispersants. Delaune et al. (1984) reported that meiofaunal densities increased in marsh plots that were flushed.

Dispersants. Mixed results of applying dispersants to saltmarsh vegetation have been reported. Baker (1971d) observed an increase in dead vegetation in marsh areas treated with dispersants relative to untreated areas. Delaune et al. (1984) reported that concentrated dispersant reduced gross CO_2 fixation in marsh plants and decreased abundances of infaunal invertebrates. Smith et al. (1984) reported oil levels in marsh sediments were the same with or without application of dispersant. *Spartina* CO_2 fixation and aboveground biomass were not affected by dispersant, and meiofaunal densities decreased after treatment with both dispersed and undispersed oil. Lane et al. (1987) observed that sensitivity to oil and oil dispersed with Corexit varied among marsh plant species, with mid-marsh vegetation in a Nova Scotia habitat being most sensitive. Vegetation located along marsh creek edges was relatively insensitive to oiling, but sensitive to dispersant, while high marsh vegetation, *Spartina patens*, was relatively tolerant of both oil and dispersed oil. Overall, in Lane et al.'s (1987) study, dispersed oil caused more injury than oil alone, with the most severe impact observed in a less well-drained mid-marsh area. Little and Scales (1987) tested the British Petroleum product, Enersperse 1037, a type III chemical dispersant consisting of a mixture of surfactant and glycol ethers in a non-aromatic solvent. Controlled experiments were conducted in a U.K. saltmarsh. Enersperse 1037 was extremely toxic to all marsh vegetation. When the dispersant was applied in combination with crude oil, the treated vegetation was almost completely destroyed. Only a few *Spartina* shoots had sprouted by the end of the growing season, and these were stunted and did not flower.

Cutting. Cutting oiled marsh plants removes oil from the marsh and prevents recontamination and continued oiling. Westree (1977) states that cutting is well-tolerated by *Spartina* marshes. Baker (1971d) reported that most vegetation in a Welsh marsh regrew within a year following cutting provided the cut area was well drained. Greater *Spartina* mortality occurred in waterlogged areas that were cut. Delaune et al. (1984) reported poor *Spartina* regrowth two years after cutting, with three full seasons were required for complete regrowth of Louisiana saltmarsh vegetation. Kiesling et al. (1988) reported that cutting removed some, but not all, oil. In cut areas, initial injury to plants was increased relative to uncut areas, as a result of the foot traffic involved in cutting operations. Complete recovery of vegetation in a Galveston Bay, Texas saltmarsh was achieved one year after cutting. Kiesling et al. (1988) recommended that cutting be conducted only when plant surfaces were heavily coated with oil which could not be flushed off. Also, cutting may worsen impacts in exposed areas because of increased potential for erosion Some recent work indicates that cutting oiled vegetation may be more deleterious than leaving the vegetation in an oiled condition (Jacquelin Michel, pers. com).

Burning. Westree (1977) recommended burning as a means of rapidly removing oil from marshes that experience winter die-back and regrowth from rhizomatous roots. Baker (1971d) reported that *Spartina* shoot densities in burned areas were not significantly different from those in unburned areas after one year. However, Kiesling et al. (1988) found that burning following oiling with No. 2 fuel oil increased the oil content of sediments in a Texas marsh and neither reduced injury nor enhanced recovery overall. In Maine, following a 1993 oil discharge, burning was performed apparently successfully. However, no follow up data are available yet upon which to base this conclusion.

No action. Westree (1977) argued that cleanup activities have the potential to cause more injury to saltmarshes than oiling in terms of aboveground plant and rhizome injury, and substrate disturbance due to foot traffic and vehicles. Because they observed no significant difference in *Spartina* biomass among all of the response treatments they examined, Delaune et al. (1984) recommended no action as the best response to oiling of south Louisiana saltmarshes. With respect to this point, Gulf coast marshes are likely to exhibit a high degree of tolerance to oil because of the high residual levels of petroleum in that environment. Kiesling et al. (1988) also recommended a no action scenario because of the significant reduction in initial plant injury relative to other response techniques, noting that considerable injury from oiling probably occurs well before the initiation of cleanup activities. Kiesling et al. (1988) noted that cleanup is particularly unwarranted in areas with good tidal flushing. Because most cleanup methods removed only some, not all, crude oil, and oil levels remained comparable to those in unoiled treatments, Kielsing et al. (1988) recommended against marsh cleanup in most crude oil discharges.

3.2.1.1.2.6 Microbiological Responses to Oiling

Microbial responses to oiling appear to depend on whether marsh sediments are toxic or anoxic. Kator and Herwig (1977) studied microbial responses to oiling in experimental enclosures in a Virginia saltmarsh. Treatments consisted of unweathered Louisiana crude oil, artificially weathered Louisiana crude oil, and an unoiled treatment to which no oil was added. Heterotrophic bacteria, fungi, chitinolytic bacteria, cellulytic bacteria and petroleum-degrading bacteria were sampled in intertidal, mid-marsh, and back-marsh areas at regular intervals for one year following treatment with oil. Basic descriptive statistics were calculated and analysis of variance was performed to determine between-treatment differences.

Mean levels of chitinolytic bacteria, cellulytic bacteria, and heterotrophic bacteria and fungi were not significantly different in oiled and control treatments over one year. Within a few days of oiling, levels of petroleum-degrading bacteria in unweathered and weathered oil treatments increased by several orders of magnitude relative to unoiled control treatments, with the differential maintained for approximately one year. Calculations based on bacterial cell mass, conversion efficiency of hydrocarbons to cell carbon, and the amount of carbon available in the discharged oil indicated that the observed duration of enrichment in petroleum-degrading bacteria could be accounted for by the volume of oil added to the marsh. The weathered oil tended to support statistically higher levels of petroleum degrading bacteria than the unweathered oil. However, this was probably because more unweathered oil was lost from the marsh due to a combination of differential volatilization and the greater mobility of unweathered oil compared to weathered oil. Weathered oil tended to adhere immediately to marsh vegetation and detritus.

Delaune et al. (1979) studied the effect of Louisiana crude oil on selected anaerobic soil processes in a Louisiana saltmarsh in controlled experiments. The details of data analysis were not reported, but it appears that analysis of variance was performed. Redox potential did not vary with crude oil addition. The biological reduction of nitrate, manganese, iron and sulphate, and the production of methane and ammonium in stirred, reduced sediments were not affected by additions of up to 10% oil on a soil-weight basis. Oil placed on the water surface caused iron, manganese and ammonium released from the sediment to the overlying water column. Delaune et al. (1979) concluded that crude oil discharged onto marsh surfaces or the surface of tidal water overlying Louisiana marshes probably has little or no influence on microbial processes because Louisiana's highly organic marsh sediments are anaerobic throughout the year. Hence, petroleum hydrocarbons had little importance as an energy source for microbial metabolism.

3.2.1.1.2.7 Bioremediation Experiments

Bioremediation consists of addition of fertilizer or other materials to contaminated environments such as oil discharge sites. This may be accompanied by tilling or other aeration activities. The goal is to accelerate natural biodegradation processes. The study of bioremediation methods as a response to oil discharges is in its infancy and no comprehensive studies of saltmarshes were located. Hoff (1992) cited two examples of bioremediation agents applied to saltmarsh environments following oil discharges. Although neither application was successful in accelerating degradation of oil, eventual development of such techniques appears promising. The two cases described by Hoff (1992) are reviewed below.

<u>Apex Barges Discharge</u>

In July 1990, a collision between three Apex barges and the tanker *Shinoussa* discharged 700,000 gallons of partially refined fuel oil into Galveston Bay, Texas. Shorelines and marshes along the northern edge of the bay were covered by oil approximately one week after the discharge. A trial application of the microbial bioremediation agent AlphaBioSea was applied to a portion of the contaminated marsh 8 days after the discharge (Mearns, 1991). Following application, the Texas Water Commission, in consultation with NOAA and the EPA, carried out a monitoring program. A premixed solution containing the microbial product and a nutrient mixture was applied with a high-pressure hose from a small boat. Samples of water and sediment were collected prior to treatment and 24, 48, and 96 hours following treatment. No differences between the treated and untreated samples were observed within 48 hours. Results from later samplings were not reported.

A number of factors may explain the observed lack of differences between treated and untreated sites. Galveston Bay is chronically affected by oil, so indigenous bacterial populations may not respond to the bioremediation product. The monitoring period may well have been too short to resolve any acceleration in oil degradation rates. In cases where enhanced microbial activity was observed following oiling, increases have usually occurred on a timescale of days to weeks. Further, the discharged oil was already partly degraded when it reached the marsh. In addition, in laboratory toxicity tests, the bioremediation product was acutely toxic to mysid shrimp but not to silversides.

Seal Beach, California Discharge

An offshore well blow-out released 400 gallons of crude oil to the atmosphere in October 1990, resulting in oiling of two to three acres of marsh grass in the Seal Beach National Wildlife Refuge. Bioremediation treatment consisted of application of the microbial product INOC 8162 and fertilizer (Miracle-Gro 30-6-6) one week after oiling, followed by application of fertilizer two weeks later. The microbial product and the fertilizer were applied by hand-spraying. Samples of unoiled, oiled and treated, and oiled and untreated grass were collected. Because no differences were observed between treated and untreated oiled marsh grass, it was concluded that the microbial product was not successful in accelerating oil degradation (Hoff, 1992).

Saltmarsh Establishment Experiments Following Oiling

Walton (1985) reported the results of a saltmarsh rebuilding experiment on Middle Line Island, a barrier island located in Great South Bay, New York. Three 3 m^2 plots located just above mean high water were sprayed with 9 liters of Arabian light crude oil. One plot each was oiled in winter, spring, and summer. Half of each plot was used as a control area, receiving no cleanup or corrective treatment. The other half was prepared for transplanting one day after the summer discharge, the last exposure to oil, by cutting *Spartina alterniflora* adjacent to the sediment surface and removing all oil-contaminated material except for the soil. One half of each cleared area was fed with slow release nitrogen and phosphorus fertilizer. Commercially produced *Spartina alterniflora* transplants were planted 16 cm apart in the prepared plots. The site was evaluated 54 days after transplanting when *Spartina* in the adjacent marsh had completed its flowering. Basic descriptive statistics were not reported, and apparently statistical analyses involving hypothesis testing were not performed. Surface density, plant height, color, and rhizome penetration were noted. Overall, fertilized transplants exhibited better survival than unfertilized transplants in all plots.

Broome et al. (1988) reviewed experiments performed to evaluate the efficacy of transplant type, fertilization, and planting season for several species of saltmarsh plants following the *Amoco Cadiz* discharge. Basic descriptive statistics were not reported and apparently statistical analyses involving hypothesis testing were not performed. *Halimione portulacoides* and *Pucinellia maritima* survived better and grew more rapidly than the other plants tested. Plug-type transplants of *P. maritima*, with 5-7 cm cores of intact root and substrate material, were superior to sprigs with no substrate material. *H. portulacoides* sprigs survived and grew well. There was considerable variation in response to fertilizer materials and rates, but both nitrogen and phosphorus were required for good transplant growth on the disturbed sites tested. At the observed rates of spread, *H. portulacoides* and *P. maritima* spaced 0.5 m apart achieved complete substrate cover in ~2 and 3 years respectively after planting. Nursery areas were established for both species, and transplants of each species were obtained within two years. Two-year-old nursery plants of *H.portulacoides* produced an average of 8 spring-type transplants and *P. maritima* produced an average of 20 plug-type transplants.

3.2.1.1.3 Non-oil Saltmarsh Restoration Studies

3.2.1.1.3.1 Salmon River Estuary, Oregon

Morlan and Frenkel (1992) described a project to rehabilitate a Pacific northwest saltmarsh located in the Salmon River estuary following 17 years of diking. Restoration efforts began in 1978 when most of the dike enclosing a 22 ha pasture was removed and tidal creeks were reconnected to the estuary. No grading, planting, or other restoration activities were performed. Monitoring began with a baseline study in 1978-1980 and continued for a total of 10 years. Rapid changes in vegetation occurred following breaching. There was a radical die-off of the upland plant species that dominated the diked pasture, accompanied by rapid recolonization by saltmarsh species carried by the tides. Thirty-one percent of the area was covered by saltmarsh plants by 1980, and 91% was covered by 1988. Initial ephemeral colonizers included saltmarsh sandspurry, dwarf alkali grass, and brass buttons, exotics that eventually disappeared from the area. Persistent native species included pickleweed (*Salicornia virginica*) and Lyngbye's sedge, which dominated the vegetation by 1988. Saltgrass (*Distichlis spicata*) was absent from the site in 1980, but became a significant component by 1984. Subsidence of the marsh surface continued to influence the recovery process during the 10 years of monitoring. Marsh surface accreted by a combination of accumulation of sediment, accumulation of organic material, and soil swelling. Because of subsidence, recovery was limited primarily to the low marsh and did not include the original high saltmarsh areas.

The project was considered successful because, with reestablishment of tidal circulation, the marsh surface began to rise slowly toward its historic elevation. The diked pasture was restored to a functioning saltmarsh containing native Pacific northwest plant species, and the reconnected tidal channels were used by numerous fish. Primary production in the restored marsh was greater than in adjacent undisturbed marshes, possibly as a result of nutrient addition from enhanced sediment input, an effect typical of young, disturbed marshes. However, the restored marsh differed from the pre-disturbance system in several respects. While the marsh surface accreted more rapidly than adjacent natural marshes, Morlan and Frenkel (1992) argued that the accretion rate was likely to diminish with time, and they estimated that recovery from subsidence would require a minimum of five decades.

3.2.1.1.3.2 Muzzi Marsh, Corte Madera, California

Tidal activity was restored to 130 acres of a 200 acre diked former marsh site on San Francisco Bay in 1980. Channels and two embayments were created around the perimeter of the site in order to enhance tidal flow to the landward portion of the marsh. Cordgrass colonized the new channels within the first year following restoration and formed dense stands over five years. Long-term changes on the marsh plain included a dramatic increase in pickleweed cover and height following channel construction. The success of the project as a restoration effort was not evaluated (Faber and Bolton, 1991).

3.2.1.1.3.3 San Francisco Bay Saltpond Number 3

A 40.4 ha diked saltwater evaporation pond was abandoned in 1965. Restoration began in 1972 when the dike surrounding the site was breached to allow tidal influx. In 1974 dredged fine-grained silty clay sediments were placed inside the dike. The following year, the dike was again breached and tidal channels were cut into the dredged material. During 1976-1977, the site was planted with sprigs of Pacific cordgrass, Pacific glasswort, and pickleweed from nearby marshes. Seeding was also attempted, but failed. The sprigs were generally successful and plant cover was visually dense by 1978, with Pacific cordgrass dominating the lower 2/3 of the site and Pacific glasswort dominating the upper 1/3. By 1986, 10 years after planting, both the upper and lower zones of the site were completely vegetated. Success of the project as a restoration effort was not evaluated (Landin et al., 1989).

3.2.1.1.3.4 Sweetwater Marsh National Wildlife Refuge California

Highway construction and excavation of a flood control channel through an existing wetland filled the entrance to Paradise Creek on San Diego Bay, California. Tidal flow was rerouted through a channel connected to the Sweetwater River. The goal of the restoration project was to create habitat for the light footed clapper rail and for the California least tern, which typically nests on nearby dredge spoil (Zedler and Langis, 1991; Zedler, 1992; National Research Council, 1992).

Restoration began in the fall of 1984 with excavation of 4.9 ha of disturbed upper intertidal marsh, including areas used previously as an urban dump. Eight lower intertidal islands and adjacent channels were constructed in the fall of 1984, and the site was planted with *Spartina foliosa* in the winter of 1985. Interplant distances were 3 and 6 ft. Transplants were fertilized with urea four times during the first year after planting. Cordgrass plants that would be destroyed by construction were salvaged from Paradise Creek and placed in a small intertidal nursery that was constructed for holding and propagation. Additional plants were moved to pots for propagation off-site (Zedler and Langis, 1991; Zedler, 1992).

Monitoring began in 1987 after three growing seasons. Three wetland functions were compared in lower marsh habitat in the constructed marsh and in adjacent natural marsh. The first, included epibenthic invertebrates, a food base for top carnivores that were one-third less abundant in the constructed marsh. The presence of less soil organic matter was suggested to explain the low densities. The second included biomass. Although the cover of transplanted vegetation expanded over five years, biomass and plant height were not equivalent in constructed and natural marshes. Shorter cordgrass provides poor cover and lacks the vertical refuge that many marsh insects require at high tide. Shorter plants in the constructed marsh were probably due to differences in nitrogen pools. The nitrogen pool was approximately 16% less in the constructed marsh, although phosphorus pools were similar. The third included nitrogen fixation, rates for which were lower on soil surface of the created marsh, apparently limited by low concentrations of organic matter. Because substrate nitrogen content did not increase over the two years it was monitored, National Research Council (1992) concluded that it was not possible to predict when the created marsh would be functionally equivalent to the adjacent natural marsh.

Because a disturbed high marsh wetland was excavated to construct the site, a net loss of wetland acreage occurred. Although cordgrass cover expanded to fill bare areas, nutrient conditions did not improve over five years. Zedler and Langis (1991) (also Zedler, 1992) constructed a "functional equivalency index" based on 11 marsh attributes including, organic matter content, pore-water organic nitrogen, surface nitrogen fixation, vascular plant biomass, foliar nitrogen concentration, vascular plant height, epibenthic invertebrate abundances, and epibenthic invertebrate species lists. On average, the constructed marsh was 60% equivalent to the adjacent marsh 4-5 years after construction.

3.2.1.1.3.5 Pine Creek, Connecticut

Pine Creek, located in Fairfield, Connecticut, drains a 2 mi^2 watershed on the north shore of Long Island Sound. The area was grid-ditched for mosquito control between 1914 and 1950, and flood control dikes were installed in the 1950s and 1960s. Saltmarsh peat was stripped, the underlying sand and gravel were excavated for highway construction, and the excavation pit was backfilled with debris and garbage. As a result, undisturbed saltmarsh was reduced from 640 to 17 acres. A large dike installed in 1969 prevented the tide from entering the marsh, but allowed drainage of rainfall and runoff. Common reed, *Phragmites,* had colonized the site, and was responsible for numerous spring and summer fires. Restoration began in 1980 with construction of a new dike with self-regulating tide gates and removal of the old dike. After 5 years, the open marsh was substantially recolonized by *Spartina alterniflora* and *Spartina patens* as well as large populations of marsh crabs and ribbed mussels. However, the original populations of breeding fish, birds, and turtle did not recolonize. Although greatly reduced, *Phragmites* was not entirely eradicated. Success of the project was not evaluated (Steinke, 1988).

3.2.1.1.3.6 Barn Island, Connecticut

Twenty ha of tidal marsh located in the Barn Island Wildlife Management Area was ditched for mosquito control in the 1930s and impounded to attract waterfowl in the 1940s, with the result that the site had developed into a *Typha*-dominated wetland. Restoration began in 1978 with installation of a culvert and removal of a flapper gate, permitting free movement of tidal waters. Vegetation transects were monitored for 12 years. Dramatic changes in the vegetation occurred during this time, with *Typha* greatly reduced overall and its distribution limited to upland marsh areas. *Spartina alterniflora* coverage increased from less than 10% before restoration to 45% in 1988. The success of the project as a restoration effort was not evaluated (Sinicrope et al., 1990).

3.2.1.1.3.7 New Jersey Meadowlands

<u>Hackensack River Meadowlands Hartz Mountain Site</u>

A 63 acre site was restored as a mitigation project for shopping center construction at Hartz Mountain, New Jersey. The project site was ditched and diked for mosquito control between 1914 and 1950. The result was that the original high saltmarsh meadow was replaced by common reed, *Phragmites australis*. An additional major change in the hydrology of the area occurred when the Oradell Dam was constructed across the Hackensack River upstream from the site. The resulting reduction in freshwater allowed greater penetration of saltwater upstream. Goals of the mitigation project were to enhance wildlife diversity and abundance by converting the site from a reed-dominated community to an intertidal saltmarsh (Berger, 1992).

Restoration actions included removal of *Phragmites* and lowering the elevation of the site to increase tidal inundation. The site was sprayed with the herbicide Rodeo by helicopter. Later hand-spraying was done to eliminate reeds. The site was shaped and graded using Priestman variable counterbalanced excavators imported from England. This earthmoving equipment has low ground pressure and is able to achieve very fine gradations in elevation. The terrain was sculpted into channels, open water, intertidal zones, and raised berms. *Spartina alterniflora* seed was planted each spring between 1986 and 1988. Detailed monitoring of the site and an adjacent area dominated by common reed was performed.

By 1991 more than 80% of the site was restored to tidal inundation, with the result that *Spartina alterniflora* was established in >75% of the lower intertidal zone. *Phragmites* did not reappear in this zone. Where reed reemerged on berms it was controlled by hand application of Rodeo. Other native marsh plants such as fleabane, rushes, and sedges invaded the site, and abundances of benthic organisms and zooplankton were similar to those in the adjacent, disturbed reed marsh. Berger (1992) considered the project successful in terms of enhancing habitat diversity, vegetative diversity, and use by birds. However, he cautioned that the project consisted of habitat enhancement and conversion rather than restoration because it did not attempt to recreate the original estuarine ecosystem that existed prior to damming.

Hackensack River Meadowlands Lyndhurst Site

A 14 acre saltmarsh in the Hackensack Meadowlands, located in Lyndhurst, New Jersey was filled during use as a dredge spoil settling basin and colonized by the common reed, *Phragmites*. Nine acres of intertidal wetlands, two acres of tidal channels, and three acres of upland terrain were created. Restoration began in the spring of 1989. Eradication of *Phragmites* was accomplished by two aerial applications of the herbicide Rodeo, a water-soluble form of Roundup. The first application killed 75% of the reeds. A second application in the fall killed the remainder. Excavation began in January 1990, with conventional earth-moving equipment operated on constructed finger roads and moveable wooden mats. Final elevations were confirmed using laser surveying equipment. Two 4-foot deep drainage channels were dug around the site. Each channel was connected to an adjacent tidal creek at the northern part of the site. One channel was also connected to the tributary of another creek at the southeast corner of the site. During June and July, peat pots of *Spartina alterniflora* were planted on 3-foot centers and fertilized with nitrogen placed in the planting holes with the pots. After one year, the marshgrass was growing, and limited reinvasion of *Phragmites* was controlled by hand-spraying individual plants. The project was considered successful (Bontje et al., 1991).

3.2.1.1.3.8 Buttermilk Sound, Georgia

A sand mound on a dredged materials island was graded to restore intertidal marsh habitat and then planted with marsh vegetation in June 1975 and May 1976. Effects of fertilization and species composition were tested. By 1982, the planted sites could not be distinguished from reference sites and by 1986 no trace of the original test plots remained in the dense vegetation. The restoration was considered highly successful by the Army Corps of Engineers (Landin et al., 1989).

3.2.1.1.3.9 Gaillard Island, Alabama

Gaillard Island is a dredged materials island constructed in lower Mobile Bay in 1980-1981 by the Army Corps of Engineers. The 52.5 ha site consists of broad, gently sloping dikes surrounding an interior containment pond. It contains a mixture of island, wetland, and aquatic habitats (Landin et al., 1989). Natural colonization by vegetation began immediately following construction. A number of plantings of *Spartina alterniflora* were made between 1981 and 1986 using a variety of low-cost techniques including plants installed in burlap plant rolls, various thicknesses of erosion control mats, grid mattresses, and anchored tires belted together across the intertidal area. The best results were obtained with plants installed in burlap plant rolls and 7.5 cm thick erosion control mats. Despite some washout of plant propagules by storms and waves, by 1986, the intertidal northwest section of the dike was stabilized. On the southern part of the dike, washout destroyed the first plantings. Replantings were partly successful and a combination of replantings and stone armor stabilized the south dike by 1987. Because of dike stabilization, the project was considered successful by the Army Corps of Engineers (Landin et al., 1989).

3.2.1.1.3.10 Southwest Pass, Louisiana

Since the mid-1970s, the Army Corps of Engineers has used unconfined dredged material placement as a means of elevating shallow bay bottoms and allowing natural regrowth of saltmarsh vegetation. One example of such marsh enhancement involved 883 ha of new intertidal deposits placed in South Pass, Louisiana, between 1970 and 1986. South Pass is a dynamic system characterized by high loss rates and subsidence. Nevertheless, by 1986, 464 ha of the site were colonized by marsh vegetation for a net gain of 408 ha over 16 years. Colonization of new plants occurred within five years, with fringes of *Spartina alterniflora* established at intertidal elevations during the first growing season. Success of the project was not evaluated (Landin et al., 1989).

3.2.1.1.3.11 Bolivar Peninsula, Texas

Goat Island in Galveston Bay was originally created from dredged material 40 years ago. Dredged material has been added from the adjacent channel as fan-shaped sandy deposits on a 3-year schedule since that time. Because of the 42 km wind fetch across Galveston Bay, severe to moderate erosion of the sandy, unconfined sediments has occurred. In 1976, the island's elevated sandy mound was graded to form a gradual slope into the intertidal zone and protected with a sandbag dike. Experimental plots were treated with various combinations of plant species and fertilizer in 1976. By 1978, *Spartina alterniflora* had spread throughout 2/3 of the lower intertidal zone and *Spartina patens* covered upper areas. By 1982, plant belowground biomass was similar to those of reference sites and above ground biomass was equal to or greater than those of reference sites. An oyster reef had formed over the sandbag dike, creating an effective breakwater. Between 1983 and 1987 oysters were harvested from the sandbag dike, compromising the dike and eliminating erosion control. As a result, portions of the marsh eroded and shoreline morphology was altered. Success of the project was not evaluated (Landin et al., 1989).

3.2.1.1.3.12 Apalachicola Bay, Florida

Drake Wilson Island in Apalachicola Bay is located on a site subject to long wind fetches conducive to erosion. In 1975 the island was enlarged by placement of silty dredged material and a wier was installed by the Army Corps of Engineers. Between 1975 and 1978, *Spartina alterniflora* and *Spartina patens* were transplanted into silty and sandy areas, respectively, from nearby donor marshes. The transplanted areas were monitored for percent survival, percent vegetative cover, seed production, stem density, biomass, and new shoot production. By September 1977, most *S. alterniflora* plots planted with dense spacing had 100% cover. Plots planted with sparse spacing had poor cover. By the end of the first year after planting, *S. patens* had achieved 75% cover. 100% cover was achieved in plots planted with dense spacing, although more sparsely spaced plants had higher growth rates. By 1982, *S. patens* had become a mixed meadow and the *S. alterniflora* marsh was well established. The effort was deemed successful by the Army Corps of Engineers, which restored the site.

3.2.1.1.3.13 Recovery of Higher Trophic Levels

Most studies of saltmarsh restoration and recovery, whether or not oiling is involved, have focused on vegetation. The exception of the West Falmouth discharge was noted above. An extensive study of benthic macrofauna in restored marshes was performed by Cammen et al. (1976a,b), who compared the benthic infaunal communities of transplanted marshes developed on dredge spoil with those of nearby natural marshes in North Carolina. Sampling was conducted over nine months in 1983. Measurements included sediment grain size, organic carbon content, sediment temperature, *Spartina* biomass, and infaunal densities. Two general patterns of infaunal development were observed. In one transplanted marsh at Drum Inlet, infauna in bare and planted areas was similar, but differed markedly from that of the adjacent natural marsh. In a second transplanted marsh at Snow's Cut, bare and planted areas had different infaunal densities, but the bare areas most resembled adjacent natural marsh. A combination of sediment characteristics and elevation differences were invoked to explain the differences in infaunal development observed between the two sites. Dredge spoil sediments at the Drum Inlet site closely resembled those of adjacent natural marshes, while those at the Snow's Cut site were finer. On the basis of organic carbon pools, Cammen et al. (1976a,b) estimated that the Drum Inlet marsh would achieve levels comparable to those of adjacent natural marshes within four years from the time of the last spoil deposit, but that the Snow's Cut marsh would require approximately 25 years to achieve such levels. When the Drum Inlet site was sampled 13 years later, infaunal densities in both created and natural marshes were considerably higher than in 1973. Infaunal densities in created and natural marshes were the same, but community composition differed dramatically (Sacco et al., 1987 cited by Moy and Levin, 1991).

Moy and Levin (1990) compared sediment properties, infaunal community composition, and *Fundulus* utilization in a created marsh and adjacent natural marshes in Dills Creek, North Carolina. Sediment organic content was lower in the created marsh than in the natural marshes. Over the three years monitored, the created marsh remained functionally different from the natural marshes. In the natural marshes, subsurface, deposit-feeding oligochaetes dominated the infauna. In contrast, in the created marsh, the infauna was dominated by tube-building, surface deposit-feeding polychates. *Fundulus* diets mirrored the observed infaunal differences. In natural marshes, diets contained more insects and detritus because oligochaetes, although abundant, were less accessible. In the created marsh, polychaetes and algae were the major dietary components. *Fundulus* abundances were markedly lower in the created marsh, probably because lower *Spartina* stem densities provided less protection from predators or fewer spawning sites.

Sacco (1994; cited by Moy and Levin, 1991) surveyed 7 pairs of natural and adjacent artificial marshes in North Carolina, ranging in age from 1-19 years. Overall, infaunal densities in planted marshes were about one-half those of natural marshes, although component organisms and proportions of trophic groups were similar in both marsh types.

Minello et al. (1987; cited by Moy and Levin, 1991) evaluated fishery species in *Spartina alterniflora* marshes created on dredge disposal sites in Texas. All sites were less than six years old. Abundances of brown shrimp, grass shrimp, pinfish and gobies were consistently statistically lower in created marshes than in adjacent natural sites.

3.2.1.1.3.14 Recovery of Saltmarsh Nutrient Pools

Craft et al. (1988) compared total nitrogen, total phosphorus and total organic carbon in the top 30 cm of sediment from natural and transplanted estuarine marshes in North Carolina. The objective of the study was to assess nutrient storage in transplanted marshes. Five transplanted marshes were sampled, ranging in age from 1-15 years, and compared to five adjacent natural marshes. Additional measurements included dry weight of macromolecular organic matter, soil bulk density, pH, humic material, and extractable phosphorus. Nutrient pools increased with increasing marsh age and hydroperiod, with the largest nitrogen, phosphorus and carbon pools observed in irregularly flooded natural marshes. Accumulation rates were greater in the irregularly flooded marshes compared to regularly flooded marshes. Pools of macroorganic matter developed relatively rapidly in transplanted marshes, approximating those of natural marshes within 10-15 years. However, development of sediment organic carbon, nitrogen and phosphorus pools required considerably longer.

3.2.1.1.4 Saltmarsh Restoration and Recovery: Summary and Conclusions

3.2.1.1.4.1 Recommended Actions

In saltmarsh habitats, the extent of injury from oiling is a function of a number of factors including geographic location, type of oil, dose of oil, amount of area affected, and season. In general, light distillates are more acutely toxic than heavier crude oils, and oil discharges that occur during winter dormant seasons cause less injury than those that occur during growing seasons. Some marshes (e.g., the Gulf coast) appear relatively tolerant of oiling, probably because of high background levels of petroleum in the environment.

<u>Response to Oiling</u>

Recommended actions following oiling of saltmarsh habitat are discussed in Section 5.2.1.1. Appropriate response and restoration actions are determined in a hierarchical fashion, depending on whether or not oil has penetrated the substrate, is adhering to the substrate, is recoverable, the vegetation is contaminated, and vegetative mortality has occurred.

It is generally agreed that response and cleanup activities in saltmarsh habitats can cause more injury than that inflicted by oiling. An often cited example is the case of the *Amoco Cadiz* discharge, in which uncleaned marshes in Brittany recovered more rapidly than those that underwent extensive cleanup (Baca et al., 1987). Hence, the minimum cleanup possible after oiling should be undertaken. If appropriate, the marsh should be allowed to recover naturally. All cleanup and response activities must be performed with care to avoid trampling the marsh substrate and plant root systems. Low pressure flushing is effective in removing oil from marsh surfaces, provided oil has not penetrated the substrate. In cases where marsh vegetation is heavily oiled to the extent that it may recontaminate the marsh, vegetation can be cut and the oiled debris removed, provided care is taken not to trample the marsh substrate or plant root systems. Replanting should be considered if recovery is slow following oiling and cleanup (i.e., including no action).

<u>Factors Affecting Success of Saltmarsh Restoration</u>

A number of physical and biological factors influence the success of restoration efforts in saltmarsh habitats including:

- Elevation;
- Wave climate;
- Topography, including slope and drainage;

- Substrate;
- Planting design and techniques;
- Trophic web considerations; and
- Human interference.

These factors are discussed separately below.

Elevation, Slope and Tidal Range

It is generally agreed that site elevation is the single most critical factor affecting the survival of emergent marine vegetation, including saltmarsh flora (e.g., Krone, 1982; Zedler, 1984; Brooks et al., 1989; Crewz and Lewis, 1991). Elevation, in combination with slope, determines the areal extent of the intertidal zone, and hence zonation of plants. Gentle slopes provide better drainage and function to increase intertidal area and dissipate wave energy over a greater area, reducing the possibility of erosion. In general, optimal planting elevations for saltmarsh vegetation at a given site are similar to their natural colonization elevations in adjacent comparable areas.

Wave Climate

Wave climate affects the initial establishment and long term stability of saltmarshes. Wave climate is described by average fetch, longest fetch, shore configuration, and sediment grain size. Planting success is inversely related to fetch. In a study of Virginia marshes, *Spartina alterniflora* and *S. patens* were established without maintenance planting at sites where the average fetch was <1.8 km. Along shorelines exposed to fetches of 1.8-6.5 km, plantings in coves and bays had a better chance of survival than those along open coasts. Maintenance planting was necessary on these types of shorelines. Where fetches were 5.6-10.2 km, marsh establishment was impractical without a permanent breakwater (Hardaway et al., 1985).

Topography and Site Design

Crewz and Lewis (1991) emphasized the value of early site preparation and planning in order to maximize timely implementation of planting efforts. In general, they recommended that wetland restoration sites have maximum contact with the marine environment and that flushing be maximized without undue wave and wind exposure. If necessary, open sites should be protected with artificial structures such as rip-rap berms. Sites should be located so as to avoid exposure to stormwater drainage from lawns and roads. However, clean stormwater can be utilized to provide flushing and a salinity gradient which promotes vegetation diversity. Slopes should be established within the minimum tidal range for the planted species, and oriented toward tidal sources. Ponding of water should be minimized by incorporating ditches, swales, and channels into the site design in order to promote drainage. Topographic complexity will usually vary with the size of the site.

Salinity

Salinity determines which species should be planted and the type of plant community that will eventually develop at a particular site. Salinities may be too high for plant growth, especially in topographic depressions that do not drain adequately at low tide.

Substrate

Grading and shaping operations are easier on sandy soils than on silt or clay because of the greater bearing capacity and traffic of sand. The low organic content and nutrient capacity of sandy substrates is a possible disadvantage, but this is not likely to be a problem where tidal waters are nutrient-rich and transport nutrient-rich sediments (Brooks et al., 1989). Most studies of the effects of fertilization of saltmarsh plants have reported ambiguous results. However, in cases where the substrate is nutrient deficient, slow release fertilizer may be applied at the time of planting and at the beginning of subsequent growing seasons, as necessary.

Sedimentation

A moderate amount of sedimentation may stimulate plant growth by providing nutrients. However, excessive sedimentation can damage plants and alter marsh elevations (Krone, 1982; Brooks, 1989). Hence, restoration sites should be located in areas with appropriate sedimentation regimes.

Planting Design and Techniques

The success of saltmarsh plantings is influenced by plant selection and planting techniques. Factors that must be considered include species composition, type and availability of planting stock, planting techniques, and spacing and density of plants. These factors are discussed separately below.

Species composition: The species composition of saltmarsh vegetation is region-specific. Marshes on the Atlantic and Gulf of Mexico coasts are dominated by *Spartina alterniflora* in lower intertidal areas and *Spartina patens* in upper intertidal areas. Pacific coast marshes tend to be dominated by pickleweed, *Salicornia virginica* in the lower intertidal and tufted hairgrass, *Deschampsia caespitosa* in upper intertidal areas, or *Spartina foliosa* (California).

Planting stock: Most published studies concern smooth cordgrass, *Spartina alterniflora*, which may be planted as seeds, bare root seedlings, sprigs and plugs. Seeds must be harvested from the field. Falco and Cali (1977), Maguire and Heuterman (1978), and Brooks (1989) reviewed methods for seed germination, storage and handling. Seeding is generally successful only in upper intertidal areas, with seeds planted in lower areas subject to washout (Seneca et al., 1976; Meeker and Nielsen, 1986). Bare root seedlings may also be subject to washout (Meeker and Nielsen, 1986). Transplantation of either nursery grown or field-dug plants may be accomplished by hand or mechanically, and is generally successful over a wider range of conditions than seeding (Seneca et al., 1976). Brooks (1989) recommended collecting field-dug plants from newer marsh environments without extensive root mats and packing them in moist sand until transplanted. Zedler and Langis (1991) established an intertidal nursery site for storage of field-collected transplants prior to transplanting. The advantages of nursery grown plants included that there is little planting shock because the intact root system is transplanted to the field and growth resumes rapidly, disturbance to natural stands is avoided, nurseries provide a source of plants when suitable digging sites are not available, and nursery grown plants can be held longer than dug plants before transplanting, if necessary. Disadvantages include cost, the need for advance planning to ensure that plants are available, and the potting medium does not contain marsh soil microfauna and microflora (Brooks, 1989).

Direct seeding of *Spartina patens* is not an option, but seedlings can be grown in pots or flats. The same transplanting techniques are used as for *S. alterniflora*. *S. patens* responds well to fertilization with nitrogen, and fertilizer can be broadcast on the soil surface (Brooks, 1989).

Spacing: Optimum spacing of *Spartina* transplants is a function of wave climate. Better survival is achieved with smaller interplant distances on exposed shores (Woodhouse et al., 1976; Broome et al., 1986). For example, Broome et al. (1986) found that 45 and 60 cm spacings were more successful in marginal sites, compared to 90 cm spacing in sites with more favorable growing conditions.

Planting methods: A number of publications provide guidance regarding planting methods. Knutson (1977) constructed planting decision keys for Atlantic, Pacific, and Gulf of Mexico coast saltmarshes. Topics covered included plant selection, planting methods, determination of seed application rates, and interplant distances, determination of fertilization requirements, and estimated labor requirements. University of North Carolina Sea Grant College Program (1981), Edwards and Woodhouse (1982), and Barnett and Crewz (1991) provide guidance for planting *Spartina alterniflora*, *S. patens* and other saltmarsh species. Topics covered include sources of plantstock, timing of planting, spacing, planting methods, and fertilization. Coultas (1980) described methods for transplanting needlerush, *Juncus roemerianus*, in Florida marshes. Zedler (1984) reviewed restoration and enhancement techniques for southern California saltmarshes. Pacific Northwest planting methods are described in Weinmann et al. (1984), Simenstad et al. (1991) and Washington State Department of Ecology (1993).

Trophic Web Considerations

Saltmarsh vegetation, whether transplanted or natural, is subject to grazing by livestock, waterfowl, and mammals. Brooks (1989) noted that Canada geese and snow geese graze cordgrass rhizomes and injure new plantings. He recommended exclusion of waterfowl by installation of wire netting on the seaward edge of planted areas. Muskrats may be excluded by trapping or fencing.

Human Interference

Human interference includes trampling, mowing, pruning, digging for bait (e.g., fiddler crabs), vehicular use, dumping and vandalism. All of these activities can impair the quality of saltmarsh wetlands. Additionally, alteration of freshwater inputs by ditching, toxic and nutrient runoff, insect spraying, domestic animal injury, and disruption of the activities of fauna (e.g., nesting, roosting, feeding) through human presence (e.g., docks, boat wake erosion) can disrupt the structure and function of saltmarshes. Crewz and Lewis (1991) recommended that sites vulnerable to public access be protected with structures that deter intrusion (e.g., signs, barriers such as fences, waterways or vegetative buffer zones). Vegetation buffer zones make sites less obvious (Zedler, 1984; Willard and Hiller, 1990). Protective structures include buffers cleared of exotic vegetation (Lewis, 1989). Such buffer zones should be maintained until the regulatory agency responsible for monitoring has determined the restoration/creation a success. Alternatively, provision of public viewing platforms or other means for the public to monitor the success of restoration efforts may counter potential negative influences of human interference.

3.2.1.1.4.2 Natural Recovery Times

Natural recovery in saltmarshes involves vegetative regrowth and reseeding of plants, with recolonization of benthic invertebrate populations by recruitment of juveniles and immigration of adults (Krebs and Tanner, 1981). Studies of salt marsh recovery from oiling have usually focused on regrowth of marsh vegetation (Johnson and Pastorak, 1985). The recovery potential of saltmarsh vegetation varies with location, oil type, oil dose, area affected, and season, ranging from 1-20 years (Booth et al., 1991). Longer recovery times may be expected in cold or otherwise limited locations. Long recovery times have been reported for marshes heavily oiled with No. 2 fuel oil, while lighter oilings and less toxic discharges have permitted faster recovery (Johnson and Pastorak, 1985). Recovery of marsh vegetation from crude oil and number 6 oil discharges has ranged from one to three years, while recovery from No. 2 fuel oil discharges has required four or more years. Recovery of benthic fauna occurs more slowly than recovery of marsh vegetation (Cammen et al., 1976a,b; Krebs and Burns, 1977; Sanders et al., 1980). Recovery of nutrient pools may occur over even longer timescales (Craft et al., 1988).

3.2.1.1.4.3 Monitoring

The importance of efficient monitoring programs following creation and restoration of wetlands was emphasized by Crewz and Lewis (1991), who noted that the need for monitoring is obvious from the injury observed at a number of the sites that they monitored. Injury includes slope erosion, encroachment from adjacent construction, debris impacts, and drainage impairments.

Crewz and Lewis (1991) recommended that monitoring begin immediately upon site restoration. Following completion of site planting, monitoring should be conducted frequently through the first six months, with quarterly, and eventually biannual, sampling conducted. Written reports and photographs should be submitted to the appropriate regulatory agency at the beginning of the project, and immediately as problems are observed. If pre-incident baseline data are not available, unoiled reference sites should be established. The oil content of saltmarsh substrate should be measured in sediment cores.

Midcourse alterations may be needed to correct problems if a site is not developing properly. For example, elevations may be inappropriate, flushing or drainage may not be adequate, or plant material may be poor. Timely mid-course alterations may correct these problems and increase the chances that the wetland will mature. Ability to correct situations through midcourse corrections can only occur if a monitoring program is in place.

Ideally, oil-affected saltmarshes should be monitored over a time period appropriate to document recovery. The timescale of monitoring will be discharge- and location-specific. As a practical matter, Crewz and Lewis (1991) recommended monitoring for a minimum of three years in saltmarsh wetlands. Monitoring over this period may be adequate for establishing short-term survival of installed plants, but longer monitoring programs, coupled with mid-course alterations, will improve the likelihood that a site matures, and should not be limited solely to plants. For example, in the Pacific Northwest, USACOE requires for permit mitigation monitoring for wetlands, but restoration monitoring has become accepted as requiring a minimum of 10 years by the natural resource trustees (NOAA; USFWS; Washington Department of Ecology; Clark, 1993).

3.2.1.1.4.4 Recommendations for Future Research

Future research needs include development of non-destructive response methods to oiling, including bioremediation, and understanding the timescale of recovery of saltmarsh functional values including nutrient pools, biomass production, faunal community development, and trophic transfers.

3.2.1.2 Mangrove Swamps

Mangrove habitats are considerably less well studied than saltmarshes. The studies reviewed concern restoration and recovery of mangrove swamps following oiling. Most published reports focus on acute impacts of discharged oil on mangroves trees (e.g., Rutzler and Sterrer, 1970). The only well studied case of chronic oiling in mangrove habitats is the Refineria Panama discharge (section 3.2.1.2.1.10). Case studies of oil discharges in mangrove habitats are reviewed in chronological order in section 3.2.1.2.1. Experimental studies of oil effects in mangrove habitats are reviewed in section 3.2.1.2.2.

3.2.1.2.1 Case Studies of Oiling in Mangrove Swamps

3.2.1.2.1.1 *Witwater* Discharge

In December 1968, the tanker *Witwater* ran aground off the Caribbean coast of Panama, discharging 20,000 barrels of diesel oil and bunker C fuel oil. Cleanup efforts consisted of removing oil from the water using unspecified methods (Birkeland et al., 1976). Injury to mangrove habitats was assessed qualitatively approximately two months after the discharge. Basic descriptive statistics were not calculated and apparently statistical analyses involving hypothesis testing were not performed. The pneumatophores of black mangroves were thickly covered with a mixture of mud and oil. Prop roots of red mangroves were coated with a thick layer of oil. Red mangrove seedlings were covered with oil and suffered massive mortality. Populations of crabs, *Uca* sp., were reduced relative to non-oiled areas. Long-term monitoring was not reported, and no estimate was made of time to complete recovery (Rutzler and Sterrer, 1970).

3.2.1.2.1.2 Tarut Bay Discharge

In April 1970, a pipeline broke on land near Tarut Bay, Saudi Arabia. A levee retained some of the oil, but 100,000 barrels of Arabian light crude oil were discharged into shallow Tarut Bay (Spooner, 1970). Restoration activities began immediately. Slicks in the bay were dispersed with Corexit 7664. Accumulations of oil along causeways were removed by a combination of road tankers, skimmers, and suction hoses. Oil that remained dispersed in the water column was removed gradually by tidal flushing. Qualitative observations were made one week and three months following the discharge. Quantitative sampling was not done and statistical analyses were not performed. Some immediate mortality of benthic fauna occurred, but some organisms survived. In dwarf mangrove (*Avicennia*) marshes, some leaves were oiled, but the substrate did not appear to be heavily oiled. After three months, some mangroves were completely defoliated, but many survived, with some bearing flowers and fruit. Spooner (1970) concluded that after three months, mangroves and associated fauna exhibited little evidence of injury.

3.2.1.2.1.3 *Zoe Colocotroni* Discharge

In March 1973, the Liberian tanker *Zoe Colocotroni* ran aground off La Parguera, Puerto Rico. In order to free the ship, approximately 4,500 tons of crude oil were pumped overboard. The wind drove about 60% of the oil into Bahia Sucia in southwestern Puerto Rico, where it affected a number of marine habitats, including red and black mangrove swamps. Response efforts were not reported nor were acute impacts of the discharge described in detail (Gilfillan et al., 1981; Nadeau and Bergquist, 1977).

Nadeau and Bergquist (1977) evaluated the discharge site and an unoiled reference site qualitatively one week, 13 weeks, and 3 years after the discharge. Statistical analyses were not performed. Observations were made of the degree of prop root oiling, of the prop root invertebrate community, and of oil in swamp sediments. They observed about half as many faunal groups on oiled prop roots one week after the discharge. Thirteen weeks after the discharge, repopulation of the prop root community began. After three years, dead mangroves were evident and oil remained in sediments.

Gilfillan et al. (1981) sampled the discharge area and an unoiled reference area in November 1978, five years after the discharge. Eleven transects in oiled areas and five transects in the unoiled area were designed to transit three subhabitats: red mangrove fringe, black mangrove areas, and a salt lagoon. Cores were collected along each transect to sample infaunal benthic communities. The reported results are qualitative. Statistical analyses were not performed. Overall, mangrove prop root communities had recovered five years after the discharge. In black mangrove areas, there were more infaunal organisms > 1 mm in size in oiled areas than in the reference sites. In red mangrove habitats, there were fewer infaunal organisms > 1 mm in size in oiled areas, reflecting the red mangrove's greater susceptibility to oiling. In the lagoon, there were higher numbers of infaunal organisms > 1 mm in size in areas that had been oiled.

Corredor et al. (1990) noted that although most petroleum released at sea in tropical environments degrades rapidly, contamination reaching intertidal sediments may persist for many years. They observed discrete subsurface layers of petroleum hydrocarbons in intertidal sediment cores collected from the discharge site in 1990, 13 years after the discharge. The uppermost such layer contained petroleum hydrocarbon concentrations greater than 200 mg g^{-1}, probably attributable to the 1977 *Zoe Colocotroni* discharge. A deeper layer with less concentrated petroleum hydrocarbons was believed to correspond to the *Argea Prima* discharge in 1962. Sediments above, between and below these layers had low concentrations of typical biogenic hydrocarbons.

3.2.1.2.1.4 *Garbis* Discharge

In July 1975, the tanker *Garbis* discharged 1,500 to 3,000 barrels of crude oil-water emulsion into the western edge of the Florida Current. Prevailing easterly winds drove the oil ashore along a 30 mile stretch of the Florida Keys from Boca Chica to Little Pine Key. Restoration efforts were not reported (Chan, 1977).

Chan (1977) compared benthic invertebrates in two oiled sites and one unoiled reference site during one year following the discharge. Basic descriptive statistics were not calculated and statistical analyses involving hypothesis testing were not performed. She observed that intertidal invertebrates were killed immediately in many mangrove fringes. Immediately following the discharge, crabs (*Uca* sp.) migrated to unoiled habitats. Snails (*Melalampus* sp.) did not ascend mangrove roots until the oil became tacky, about 4 weeks after the discharge. Red mangroves with >50% of their leaves oiled were killed, and red mangrove propagules with >50% oil coverage died within two months. Black mangroves with >50% of pneumatophores oiled were killed. Thin oil coating left chemical burn scars and germination of oiled seeds decreased by 30%. In mangrove swamp/*Batis* marsh habitats, all epifaunal organisms died immediately in heavily oiled areas. *Batis* and *Salicornia* spp. died when oil coated their leaves, stems, or substrate. Lightly oiled mangrove areas appeared to exhibit normal growth six months after the discharge. However, young and dwarf mangroves apparently suffered permanent injury, indicated by deformed leaves, roots and stems.

Getter et al. (1981) visited the discharge site in May 1980, five years after the discharge, and reported that the oil had weathered significantly. Although aerial and ground surveys of oiled and reference sites were performed, Getter et al. (1981) did not report the results, stating only that weathering of the oil made statistical comparisons difficult. The time to complete recovery was not estimated.

3.2.1.2.1.5 *St. Peter* Discharge

In early February 1976, the Liberian tanker *St. Peter*, carrying a cargo of 243,000 barrels of Orito crude oil, sank in 1,000 m of water about 30 km off Cabo Manglares, Colombia. By mid-February, oil slicks reached mangrove habitats in Colombia. Mangrove roots and trunks located 20-70 m from the shoreline were oiled to heights of 2-3 m. Response efforts were not reported. Mangrove trees in the impacted area were partly defoliated and massive invertebrate mortality occurred: mangrove barnacles, mussels, and oysters were rare or absent two months after the discharge. Motile invertebrates migrated out of the affected area to zones above the oil line. Fiddler crab populations were reduced, particularly younger life history stages (Hayes, 1977; Jernelov and Linden, 1983).

Response efforts were not undertaken due to lack of equipment (Hayes, 1977). The discharge site was monitored, using methods that were not reported, in May and June 1976, 3-4 months after the discharge. Basic descriptive statistics were not reported and statistical analyses involving hypothesis testing were not performed. By this time, most of the oil had washed off of the roots and trunks naturally in the less heavily oiled areas. New mangrove leaves, blooms, and seedlings were present in previously defoliated areas, and most crustaceans and molluscs had returned to prespill levels, presumably by migrating from unoiled areas (Jernelov and Linden, 1983). The time to complete r

3.2.1.2.1.6 Ensenada Honda Jet Fuel Discharge

In November 1976, jet propulsion fuel (JP-5) leaked from a storage tank, flooded a catchment basin, and discharged into Ensenada Honda, Puerto Rico, where 59,000 gallons of it collected in two mangrove forest areas. No response activities were undertaken. One of the affected areas, a mixed species assemblage of red, black, and white mangroves, was surveyed 152 days and 328 days after the discharge. Three transects were monitored in the oiled area and a single transect was monitored in an adjacent unoiled reference area. Detailed measurements were made in $10m^2$ quadrants along the transects. Adult trees were identified to species and categorized as dead or alive. Tree height, diameter, and canopy cover were measured. Seedlings were counted and marked. Invertebrate fauna were enumerated. Sediment and water samples were collected for analysis of petroleum hydrocarbons. Basic descriptive statistics were not reported and statistical analyses involving hypothesis testing were not performed (Ballou and Lewis, 1989).

Aerial surveys revealed that immediately following the discharge, 5.5 ha of mangrove forest were completely defoliated and 0.8 ha were partially defoliated. There were also extensive injuries in tidal creek forest north of the principal impacted area. Seedling mortality was variable among the oiled transect stations and appeared to be correlated with degree of exposure to open water. Petroleum hydrocarbons were not detectable in water samples collected 152 days and 328 days after the discharge. Sediment samples collected at the same time contained low levels of residual hydrocarbons. Ballou and Lewis (1989) concluded that the mechanism of toxicity was direct poisoning of mangroves by the jet fuel. They proposed that recolonization of the affected mangrove forest depends on an adequate supply of new seeds in combination with acceptable growing conditions. Seeds were available from adjacent unaffected areas and colonization was evident about one year after the discharge. Cleanup was not recommended because the highly volatile jet fuel evaporated rapidly, leaving low residual petroleum hydrocarbon concentrations. Removal of dead trees was not recommended on the grounds that it was likely to injure the recolonizing seedlings. A 10-year recovery was predicted under the natural recovery scenario.

3.2.1.2.1.7 The *Howard Star* Discharge

In October 1978 the ship *Howard Star* released ~40,000 gallons of Bunker C and lubricating oils into Tampa Bay, Florida. At least 20 km of fringe mangrove shoreline were affected. Response efforts were not reported (Getter et al., 1981).

Getter et al. (1981) visited oiled sites in Tampa Bay 2 months, 9 months, 14 months, and 16 months after the discharge. Each discharge site and adjacent reference sites were examined by aerial surveys to locate defoliated areas. Areas with obvious defoliation and reference areas were selected for subsequent ground surveys. Oil impacts were assessed by comparing ecological parameters at oiled and reference stations using a statistical "compartmental method" to test the null hypothesis that no significant biological changes were induced in defoliated areas by the oil discharge. Significant parameters were then examined in transects located along a degree-of-oiling gradient. The heaviest defoliation of mangroves, seedling mortalities, and mortalities of canopy-dwelling animals were observed where the heaviest oiling had occurred. The degree of oiling was controlled largely by geomorphic features of the forest.

On the basis of geomorphic features, two types of oil impacts were observed in Tampa Bay: outer fringe and an inner basin impacts. Impact to the outer fringe occurred at two sites where defoliation was concentrated in the outer mangroves. In these areas, mangrove mortality appeared to be related to degree of exposure to waves and currents and degree of oil penetration into the forest substrate. The latter was enhanced by the presence or absence of burrowing crabs. In Tampa Bay, exposed areas contained few burrows and oil was removed by wave action within a few weeks. Impact to the inner basin was observed in two oiled areas of Tampa Bay where high tides moved oil up over coastal berms and into shallow basins behind them, spreading the oil over a wide area with a less well defined effects (Getter et al., 1981). Time to recovery was not estimated.

3.2.1.2.1.8 The *Peck Slip* Discharge

In December 1978 the barge *Peck Slip* released 440,000-460,000 gallons of Bunker C fuel oil into Bahia Medio Mundo, Puerto Rico, oiling at least 10 km of mangrove-dominated shoreline. Cleanup efforts were not reported (Getter et al., 1981). Getter et al. (1981) visited oiled sites in Media Mundo immediately after the discharge, and 3-4 months and 10 months after the discharge. As in the *Howard Star* discharge, each discharge site and adjacent reference sites were examined by aerial surveys to locate defoliated areas. Areas with significant defoliation and reference areas were selected for subsequent ground surveys. Oil impacts were assessed by comparing ecological parameters at oiled and reference stations using a statistical "compartmental method" to test the null hypothesis that no significant biological changes were induced in defoliated areas by the oil discharge. Statistically significant parameters were then examined in transects located along a degree-of-oiling gradient. The heaviest defoliation of mangroves, seedling mortalities, and mortalities of canopy-dwelling animals were observed where the heaviest oiling had occurred. The degree of oiling was controlled largely by geomorphic features of the forest.

On the basis of geomorphic features, two areas of oil impact were observed at Media Mundo, an inner fringe impact and an inner basin impact. In the inner fringe impact, oil was concentrated on the inner mangroves, which are located on the inner berm of the forest. The affected inner berm site became heavily defoliated within two months of oiling and remained so 18 months later, with the substrate and prop roots remaining oiled even after Hurricane David in 1979. An inner basin impact, similar to that described in Tampa Bay, was also observed at Media Mundo. Time to recovery was not estimated (Getter et al., 1981).

3.2.1.2.1.9 Northern Red Sea Discharge

South Geisum Island in the northern Red Sea was heavily oiled by a series of discharges from undefined sources during 1982 and 1983. The volume of oil discharged was not reported. The oil, which was viscous and weathered, completely coated the pneumatophores of *Avicennia marina*. However, most trees survived. Dicks and Westwood (1987) investigated the reason for survival of the heavily oiled trees. Preliminary observations suggested that the surviving trees had low densities of breathing roots and inhabited coarse, well drained sediments. In contrast, dead trees inhabited muddier sediments and had higher densities of breathing roots. Thus, sediment characteristics were investigated in greater detail in the field in areas where oiled mangroves had survived, where oiled mangroves had died, and in areas containing unoiled mangroves. Measurements were made of soil redox potential, salinity and oxygen content of interstitial water, sediment hydraulic conductivity, breathing root density, and oil layer thickness. The particle size, hydrocarbon content, and infaunal biota of sediment cores were characterized. Dicks and Westwood (1987) concluded that oiled mangroves survived in well drained sediments and died in muddy, poorly drained sediments. Survival was related to the number of breathing roots. Mangroves inhabiting muddy sediments had extremely high densities of breathing roots, probably in response to anoxic conditions in these sediments.

3.2.1.2.1.10 The Refineria Panama Discharge

On April 27, 1986 a storage tank at the Texaco Refineria Panama on the Caribbean coast of Panama ruptured, releasing ~240,000 barrels of medium weight crude oil into Cativa Bay. Most of the oil was held by containment booms for 6 days while it was removed by skimmers and shore-based pump trucks. On May 3, aircraft sprayed 21,000 liters of the dispersant Corexit 9527 onto the oil slicks. This action was considered ineffective because the dispersant was deployed a week after the discharge when oil had already weathered and because seas were calm at the time of spraying. On May 4, a storm broke the containment booms, releasing ~150,000 barrels of oil into the Atlantic Ocean. Winds, tides, and rain runoff washed part of the oil onto exposed shorelines. Some of the oil was carried back into Cativa Bay and some was washed into adjacent embayments with mangrove shorelines. By May 15, oil had spread along the coast and washed across fringing reefs and into mangrove forests and small estuaries within 10 km of the refinery. Channels were dug through mangrove areas to drain oil, but appeared to increase the shoreward movement of oil. Physical disturbance by workers digging the channels appeared to increase subsequent erosion.

A total of 82 km of coastline (=11 linear km) was oiled to varying degrees. Approximately 75 ha of mangroves, primarily the red mangrove *Rhizophora mangle*, were killed by the discharge. Severe mortality of oysters and other invertebrates inhabiting mangrove roots was reported (Cubit et al., 1987; Jackson et al., 1989; Teas et al., 1989a; 1989b; Keller and Jackson, 1991). Oil slicks were observed frequently in Bahia Les Minas during the four years following the discharge. The slicks appeared to originate primarily from fringing mangrove areas that had been impacted by the discharge. As dead red mangroves decayed and their wooden structures disappeared, erosion of the associated oiled sediment occurred, releasing trapped oil (Keller and Jackson, 1991).

The discharge site is located near the Smithsonian Tropical Research Institute, in the same area as the 1968 *Witwater* discharge. Pre-incident data on organismal distribution and abundance were available. In mangrove habitats, the discharge site was monitored between 1986 and 1992. Oiled and unoiled areas of three habitat types were monitored: open coast, lagoon, and river, for a total of 26 study sites, with replication. Aerial surveys and ground transects were performed. The focus was on the red mangrove, which was most heavily impacted by the discharge. Trees were identified to species, and height, girth, and inter-tree distance measured. Primary production was measured, various parameters of seedling demography, growth and recruitment were determined, and seedling growth was measured in transplant experiments. Seedling growth was followed by enumerating nodes (leaf scars) on vertical stems. Basic descriptive statistics were calculated for all parameters, and tests of significance were performed. Keller and Jackson (1991) reported preliminary results of the long-term monitoring. Three years after the discharge, there were no statistically significant differences in rates of leaf production and net canopy production in oiled and unoiled habitats.

Because a number of seedlings survived the discharge while adult trees died, it was concluded that adult mangrove mortality was the result of suffocation rather than oil toxicity. Their morphology (lack of prop roots) apparently allowed seedlings to survive immersion in oil. Keller and Jackson (1991) noted that, in addition to direct mortality, oil altered the physical structure of the mangrove habitat. Defoliation removed the weight of leaves from mangrove branches. In some cases, branches flexed upward, lifting roots out of the water, with the result that root-living organisms that had survived oiling then died of desiccation or heat stress. Keller and Jackson (1991) reported that the number of post-discharge recruits appeared to be sufficient for reforestation of the oil-impacted habitats. Three years after the discharge, dense growths of young seedlings were observed. Some of these were natural recruits and some had been planted (described in section 3.2.1.2.2 below). Garrity et al. (1993a; 1993b) reported significant reductions in the total length of shoreline fringed by red mangroves five years after the discharge. In areas where mangroves survived or regenerated, submerged prop roots, an essential habitat for biota, were fewer in number and extended less deeply into the water than before the discharge. Oysters and mussels collected between 1986 and 1991 had high tissue levels of hydrocarbon residues associated with reduced population levels during the same period.

Keller and Jackson (1991) also reported effects of the discharge on invertebrate communities inhabiting mangrove prop roots. Prespill data were available from studies conducted by the Smithsonian Tropical Research Institute in 1981 and 1982. Quarterly post-discharge monitoring began in August 1986, four months after the discharge. Quantitative surveys of oiled and unoiled areas of three intertidal habitats were surveyed: mangroves fronting on the open ocean, mangroves located along channel banks and lagoons, and mangroves located along brackish streams and man-made ditches. Basic descriptive statistics were calculated for all measured parameters and statistical testing was done. Oil was present in mangrove sediments and continued to appear on mangrove roots during the three years following the discharge, with the highest levels of continued oiling occurring in stream habitats and the lowest levels along the open coast. Rates of root mortality were 31%, 71%, and 58% in oiled open coast, channel and stream sites respectively. The same rates in unoiled sites were 2%, 2% and 4%. Open coastal habitats exhibited persistent effects of oiling after three years: abundances of the prespill dominant crustose and foilose algae were reduced on oiled roots. Distributions of sessile invertebrates were negatively correlated with the presence of oil, with the exception of the high intertidal barnacle *Chthamalus* sp. Mangrove root communities in channel and lagoon habitats also showed effects of oiling 3 years after the discharge. Before the discharge, root communities in these areas were dominated by the edible oyster *Crassostrea rhizophorae* and the barnacle *Balanus improvisus.* Abundances were lower after the discharge, with little evidence of recruitment, although oyster cover increased gradually on oiled roots. Mangrove root communities in drainage habitats were the most severely impacted by the discharge. The discharge completely eliminated the mussel *Mytilopsis sallei*, which dominated root communities in these habitats. Less common epibionts were also eliminated. Three years after the discharge, the root systems continued to be reoiled, and there was no evidence of recruitment of mussels or other epifauna.

Garrity et al. (1993b) monitored the epibiota on red mangrove prop roots for five years following the discharge. Prop roo communities in three habitats were followed: wave-washed open shores, channels and lagoons, and interior drainage systems. Measurements included release of weathered oil, dissolved and suspended hydrocarbon concentrations, mangrove root areas, and abundance of mangrove root biota. The extent of structural damage to the mangrove forest was also evaluated. Extensive statistical analyses were performed. The epibiota of submerged mangrove roots did not recover completely in any habitat after five years. The structure of the mangrove fringe changed significantly after oiling. The amount of shoreline fringed with mangroves decreased, with concomitant decreases in the density and sizes of submerged prop roots. Overall, the surface area of submerged mangrove roots decreased by 33% on the open coast, 38% in channels and 74% in streams.

Initial weathering removed labile oil components such as n-alkane hydrocarbons from oiled surface sediments within six months after the discharge. However, total oil concentrations remained high, up to 20% of dry weight in surface sediments, for at least the first four years following the discharge. Residual pools of oil in mangrove sediments were sufficiently fluid to flow out when sediments were cored or disturbed five years after the discharge. Most of the oozing oil was highly degraded, but one oiled stream contained a fresh oil residue with a full suite of n-alkanes. Subsequent chemical analysis confirmed that this oil was the crude oil mixture discharged in 1986 (Burns et al., 1993). Release of oil from pools under and around the collapsed Refineria Panama storage tank and from mangrove sediments caused chronic reoiling for at least five years following the discharge, and undegraded oil residues were found in some heavily oiled sediments six years after the discharge (Burns et al., 1993; Garrity et al., 1993a). Thus, the discharge site, initially injured by a single point-source of oil, became a chronic source of oil contamination. Hydrocarbon chemistry confirmed the long-term persistence of crude oil in mangrove sediments, with pools of trapped oil maintaining consistent hydrocarbon composition. The frequency and amount of reoiling differed among habitats. Secondary reoiling was heaviest in sheltered drainage systems of the mangrove environment, where oil continuously leaked from the sediment, but also occurred along the open coast and along channels. Seasonal variation in weather, water levels and tidal flushing affected the amount of oil released. The greatest amount of reoiling occurred between February and August 1989 and appeared to be related to the collapse and cutting of dead mangroves and to replanting efforts by the Refineria Panama. Burns et al. (1993) suggested that the amount of oil released may have begun to decline at the time the monitoring program was terminated five years after the discharge, as mangroves became reestablished at oiled sites and developed root mats able to stabilize the substrate.

In contrast, bivalves in channels and streams accumulated water soluble fractions of crude oil between 1986 and 1991 and remained heavily contaminated in May 1991, five years after the discharge. Levels of suspended oil after five years were high enough to reduce bivalve growth and respiratory rates. Oysters consistently accumulated about half as much total oil as mussels (Burns et al., 1993). Erosion is thought to be the principal process releasing tarry oils from sediments, while a combination of erosion and diffusion releases suspended oils from sediments. Burns et al. (1993) suggested that the observed continued high bivalve tissue concentrations of oil were indicative of dissolved and suspended hydrocarbons in the environment declining more slowly than visible, tarry residues and proposed that the processes controlling the two types of residue were partially uncoupled. Garrity et al. (1993b) concluded that the combination of chronic reoiling, injury to epibiotic assemblages, and reductions in submerged prop root substrate had decreased productivity in the mangrove habitat. They suggested that recovery would be a complex and prolonged process, and that reductions in productivity caused by oiling would persist until the amount of submerged prop root substrate returned to prespill levels.

3.2.1.2.2 Experimental Studies of Oiling in Mangrove Swamps

Few experimental studies of mangroves have been done in the context of recovery or restoration after oiling. Getter et al. (1983) performed experimental studies to determine the effects of oil and dispersants on seedlings of red and black mangroves. Seedling stocks were collected from one site in Florida and 5 sites in Puerto Rico. Bunker C oil, No. 2 fuel oil, and light Arabian crude oil with and without the dispersant Corexit 9527 were tested. Oil doses were 25, 50, 500, 5,000 and 50,000 ppm. Dispersant concentrate was added in a 1:22 ratio. Oil was applied by injection into the root system of each plant over a period of 10 weeks. Control treatments were injections of distilled water. New leaf area and leaf shapes were monitored. Analysis of variance was performed to determine between-treatment differences. For all treatments, black mangroves were more sensitive than red mangroves, with threshold doses of 5,000 ml/L and 50,000 ml/L respectively. Lighter petroleum substances (Arabian crude oil, No. 2 fuel oil) were the most toxic, while bunker C oil was the least toxic of the oils tested. Dispersant alone and undispersed light Arabian crude oil had the greatest effect on leaves in both mangrove species. Dispersed bunker C oil was less toxic than bunker C oil alone; with other oils, dispersant increased foliage loss. Red mangroves collected from chronically oiled areas showed significant resistance to oiling with undispersed Arabian crude oil. No resistance occurred with dispersed light Arabian crude oil or dispersant alone. The authors concluded that each oil should be considered separately when the use of dispersants is considered. Because of their greater resistance to oiling, Getter et al. (1983) recommended that red mangroves be used in restoration efforts when environmental factors (flushing, salinity) are appropriate.

Lai and Feng (1985) evaluated the toxicity of dispersed and undispersed oil to *Avicennia* and *Rhizophora* species in Malaysia under conditions of varying water flow rate in the laboratory and in the field. Light Arabian crude oil was tested in laboratory experiments. Light Arabian crude oil, Malaysian crude oil, and Bunker C oil were tested in field experiments. Oil doses were 0.01-1.2 ml cm^{-2} in laboratory experiments and 0.005-0.5 ml cm^{-2} in field experiments. In treatments where oil was dispersed, Corexit 9527 was used in a ratio of 1:20 with oil. Treatments were replicated, and descriptive statistics, LD_{50}s, and cumulative percent mortalities were calculated. Dispersed and undispersed oil were both toxic to both mangrove species under static and semi-static flow conditions. Undispersed oil resulted in a slight increase in acute toxicity compared to dispersed oil. Mortality decreased more rapidly with increased water circulation due to flushing out of emulsified oil particles. Oil effects on saplings were related to smothering of the root system and lenticels by the aromatic fraction of the oil. Light Arabian and Malaysian crude oils, which contain more aromatics than Bunker C oil, were more toxic to saplings than Bunker C oil. Most mangrove mortality was attributed to passive surface deposition of oil, as well as to active uptake. Gas chromatographic analysis showed that leaf tissue was the principal accumulation site due to active uptake.

Teas et al. (1987) compared the effects of dispersed and undispersed south Louisiana crude oil in enclosed 3 m^2 plots of *Rhizophora* mangroves near Miami, Florida. The oil dose was 38 L m^2. An unnamed non-ionic water-based dispersant was used. Mangroves were oiled, then subsequently treated with dispersant and seawater and seawater alone. Treatments were applied to the mangrove plots by high pressure spraying. Analysis of variance was performed to determine between treatment differences. Results showed that south Louisiana crude oil killed *Rhizophora* mangroves. High pressure washes with seawater or dispersant one day after oiling did not reverse toxicity, and dispersant was not more toxic than seawater alone. It should be noted that the effects of high pressure washing were not evaluated as an experimental treatment.

Teas et al. (1989a; 1989b; 1991) tested a number of planting methods for mangrove propagules in a post-discharge environment where the soil still contained oil following the Texaco Refineria Panama discharge in April 1986. Their ultimate goal was to develop techniques for the rapid restoration of mangrove forests. The short-term goal of their studies was to determine when the oiled soil in the injured mangrove forests was suitable for replanting. Replanting experiments began about three months after the discharge. Propagule survival and production were measured in plantings initiated at oiled sites at three month intervals over one year. Statistical analyses were performed. At oiled sites, propagules planted three and six months after the discharge failed to develop roots and all died. Mortality began to decline in propagules planted nine months after the discharge. Six months after planting, survival of propagules planted at oiled and unoiled sites did not differ significantly. Propagules planted at oiled sites in holes filled with unoiled soil grew more rapidly than those planted in holes filled with oiled soil. Mangrove seedlings planted in oiled soil appeared to be less sensitive to oil than propagules (Teas et al., 1989a). Seedlings planted directly in upland soil in holes lined with plastic foam to exclude oil grew better than seedlings planted directly in oiled soil, in upland soil with other types of plastic liners, or in upland soil with dispersant added (Teas et al., 1989b).

Seedling development was enhanced by planting propagules in cylinders of upland soil rather than directly in oiled mangrove forest soil 28 months after oiling. The oiled soil was neither toxic nor nutrient deficient, but it was dense and peaty, and did not support vigorous *Rhizophora* growth (Teas et al., 1991).

Field plantings of nursery seedlings and propagules in oiled areas were initiated 12 months after the discharge. Because there was no significant difference, except in the number of prop roots (a function of plant age), in growth rates of nursery seedlings and propagules, Teas et al. (1989a) concluded that nursery seedlings were not required for mangrove replanting. Large-scale field plantings in oiled sites in Panama were undertaken 12 months following the discharge. Approximately 42,000 nursery plants and 44,000 propagules were planted in holes dug in the local sandy clay soil. Plants were fed with slow release fertilizer. Initial inter-plant distance was 60 cm, later changed to 1-2m. Teas et al. (1989a) reported >90% survival after 8-10 months. Longer term monitoring data were not reported. Time to recovery was not estimated, although Odum et al.'s (1975) figure of 20 years until mangrove maturity was cited. Levings et al. (1993) reported that felling of dead mangroves, trampling, and especially digging associated with replanting activities by the Refineria Panama disturbed sediments saturated with oil. Based on the number of propagules or young trees planted and the reported size of the holes dug, Levings et al. (1993) estimated at least 340 m^2 of oiled sediments were dug and left lying on the surface of mangrove habitat, where they exacerbated reoiling. Embedded, dead roots acted as oil conduits from deep sediments to the surface. Dead and cut trees were moved by wind and water, knocking over seedlings and small trees (S.C. Levings, personal communication).

Balloetal. (1989) performed a long-term experimental study on the Caribbean coast of Panama to evaluate the use of chemical dispersants to reduce the adverse environmental effects of oil discharges in nearshore tropical waters. Two sites were monitored before, during, and after simulation of an unusually high, worst case discharge of dispersed Prudhoe Bay crude oil and a moderate level of undispersed Prudhoe Bay crude oil. A third site served as an unoiled control area. Doses were 50 ppm of dispersed oil over 24 hours, equivalent to 1,200 ppm-hours and 1 liter m^2 of untreated oil, equivalent to a 100-1,000 barrel discharge. Statistical analyses were performed. Overall, more oil was present in mangrove sediments in areas where oil was not dispersed. The canopy coverage of adult mangroves did not change appreciably at the site treated with dispersed oil, but decreased dramatically at the site treated with undispersed oil. Most defoliation occurred within four months of oiling. Three groups of red mangrove propagules were planted at each site immediately before oiling. As with adult mangroves, undispersed oil exerted more negative effects than dispersed oil. Increased numbers of invading seedlings were observed colonizing clear areas at the site treated with undispersed oil one year after oiling. Fewer seedlings had recolonized the area treated with dispersed oil, which had less clear area. Time to recovery was not estimated.

3.2.1.2.3 Mangrove Recovery and Restoration: Summary and Conclusions

3.2.1.2.3.1 Restoration of Unoiled Mangrove Swamps

The following discussion concerning the creation and restoration of mangrove wetlands encompasses general background information and guidelines, with the understanding that the details of any given mangrove wetlands creation or restoration project will be case- and site-specific. Factors influencing the overall success of mangrove restoration projects are reviewed by Crewz and Lewis (1991) and Citron-Molero (1992). These factors may be divided into four broad categories: design and planning, construction techniques, planting techniques, and monitoring and regulatory review. The first three factors are discussed in section 3.2.1.2.3.1.1. Monitoring and regulatory review are discussed in section 3.2.1.2.3.4.

A number of physical and biological factors affect the success of restoration efforts in mangrove habitats:

- Elevation;
- Wave climate;
- Topography, including slope and drainage;
- Substrate;
- Planting rationale and techniques;
- Trophic web considerations, including predation; and
- Human interference.

These factors are discussed separately below.

Elevation

It is generally agreed that planting elevation is the single most critical factor affecting the survival of emergent marine vegetation, including mangroves (Lewis, 1989; Crewz and Lewis, 1991; Citron-Molero, 1992). Optimal elevations are species specific. They are also likely to be location specific, reflecting the influences of tidal cycles, which operate on daily, monthly, and annual time scales and of local topography (e.g., the influence of channels, ditches, swales and ponds). In general, acceptable planting elevations for mangroves at a given site are similar to their natural colonization elevations in adjacent comparable areas.

Wave Regime

Characteristically, mangrove forests develop on low energy shorelines and appropriate wave climate is critical. Seedlings, propagules, and young trees may be removed directly by waves or by wrack (i.e., organic, primarily plant, material stranded on shorelines above the tide) and debris moved by waves (Teas et al., 1975; Teas, 1977; Goforth and Thomas, 1979; Lewis, 1989). Even at otherwise sheltered sites, wave energy in the form of boat wakes may remove or inhibit colonizing mangroves (Hannan, 1975).

Topography and Site Design

Crewz and Lewis (1991) and Citron-Molero (1992) emphasize the value of early site preparation planning in order to maximize timely implementation of planting efforts. In general, they recommended that wetland restoration sites have maximum contact with the marine environment and flushing be maximized without undue wave and wind exposure. If necessary, open sites should be protected with artificial structures such as rip-rap berms. Sites should be located so as to avoid major stormwater drainage from lawns and roads.

Slopes should be established within the optimum tidal range for the planted species and should be oriented toward tidal sources. Because steep slopes provide less area within an appropriate tidal range, slopes should be as gradual as possible. Steep slopes are characteristic of disturbed wetland habitats and thus are at risk of invasion by exotic vegetation. Gentle slopes have the potential to function as buffer zones, encouraging colonization and growth of saline-adapted vegetation and inhibiting invasion by exotics. Ponding of water should be minimized by incorporating ditches, swales, and channels into the site design in order to promote drainage. Topographic complexity will usually vary with the size of the site (Crewz and Lewis, 1991).

Substrate

Suitable substrate is necessary for successful restoration of mangrove habitat. If hard substrates are not sufficiently porous, plant roots tend to remain in the planting hole, stunting growth (Crewz and Lewis, 1991). Clay layers are generally anoxic. Sand is subject to erosion and lacks an organic matrix. Substrate characteristics can be assessed in advance of planting activities by taking soil cores. If substrate characteristics are unsuitable, the site may not be appropriate for planting without substantial alteration. Physico-chemical characteristics, such as soil texture, nutrient and organic content, pH, etc., can affect plant growth as well as microbial and animal populations important to habitat quality. Depending on its characteristics, the substrate may be modified prior to planting in terms of pH (addition of calcareous material to buffer acidic substrates) or nutrients. Crewz and Lewis (1991) note that, although plants may respond positively to small amounts of fertilizer at planting (e.g., Goforth and Williams, 1983), acclimation to long-term conditions is more desirable. Reark (1982) reported that fertilization was necessary to grow *Rhizophora* in beach sand. Savage (1978) recommended use of marine wrack as fertilizer for mangrove plantings.

Planting Rationale and Techniques

The success of mangrove plantings is influenced by plant selection and planting techniques. Factors that must be considered include species composition, type and availability of planting stock, planting techniques, and spacing and density of plants. These factors are discussed separately below.

Species composition: Understanding the species successional patterns of wetland vegetation is important. The ultimate target species of wetland creation or restoration efforts may not be the initial colonizer in natural situations (Lewis, 1981). "Nurse" species, such as *Spartina*, may be appropriate initial plantings for mangrove restoration projects. Lewis and Dustan (1975) observed that red, white, and black mangroves occurred within older, central areas of *Spartina alterniflora* stands in a number of sites in southern Florida. Shading and elimination of *S. alterniflora* by larger mangroves was observed. Lewis and Dustan (1975) suggested that, in these areas, natural succession progressed from *Spartina* to mangroves. They suggested a mangrove restoration strategy of initial planting of *Spartina* in created mangrove wetlands. Under this scheme, mangroves would be planted later, after *Spartina* establishment, or else would be allowed to recolonize naturally. Most mangrove wetland creation and restoration projects have used monospecific plantings of red mangroves. Because red mangroves do not develop extensive root mats, Teas (1981) recommended planting mixtures of red mangroves with black and white mangroves, which develop such root mats.

Planting stock: Mangrove planting stock includes wild and nurseried propagules and seedlings, wild and nurseried small trees, and wild large trees (Lewis, 1982). Planting techniques include direct planting of propagules, aerial planting of propagules, transplanting seedlings, small trees and large trees (Teas, 1977; Teas et al., 1978; Lewis, 1982). Success of planting varies widely depending on the type of plant material used, techniques of planting, and suitability of the planting site (see above). It should be noted that transplanting trees from the wild may be destructive of extant mangrove habitat, depending on the methods used (Hoffman and Rodgers, 1981). However, Pulver (1975) argued that within the time required for a 1.0 m mangrove tree to grow to 1.8 m, at least 50% of the trees will be naturally thinned out as a result of competitive interactions. He suggested that, in theory, every other tree will be available for transplanting. Crewz and Lewis (1991) argued against the use of field-collected mangrove trees as overly destructive of the environment. Nurseried mangroves ranging in age from one to five years are readily available. One- to two-year old red mangrove seedlings have been used in most mangrove plantings in Florida (Creuz and Lewis, 1991). Plant characteristics, availability, planting guidelines, and maintenance guidelines for 17 species of salt tolerant plants, including red, white and black mangroves, are summarized by Barnett and Crewz (1991).

Lewis (1989) recommended using local plants as much as possible, and considering natural invasion by "volunteers" from adjacent sites, where available (Lewis, 1989). For example, Sherrod (1986) reported that transplantation of *Rhizophora mangle* from northeast Florida to the Texas coast was ultimately unsuccessful because the mangroves were not able to survive freezing conditions in Texas that exceeded their physiological tolerance.

Spacing: Between-plant distances should approximate natural recruitment densities for the area (Crewz and Lewis, 1991). Citron-Molero (1992) suggests 1.0 m spacing as a balance between economy and rapid full cover. Closer spacing may be needed to compensate for erosion losses.

Planting methods: Planting methods for rooted and unrooted red mangrove propagules and red mangrove seedlings are described by Crewz and Lewis (1991) and Citron-Molero (1992). Unrooted red mangrove propagules are collected easily in season (late summer and fall) from natural populations. Costs are low and direct installation of red mangrove propagules has a number of advantages over installation of rooted seedlings. Because propagules are not expensive, greater planting densities are possible. Because field-collected propagules have not been influenced by nursery conditions, they adapt more readily to the habitat in which they are planted. Further, because propagules are not top heavy like many potted seedlings, they are less subject to wind damage.

Staked, rooted seedlings may be more appropriate to plant in less stable sites where movement of water and sediment can uproot propagules. Rooted mangrove seedlings can be planted at slightly lower elevations than propagules because of better transpiration by leaves on rooted seedlings, and rooted seedlings provide greater short-term plant cover than propagules. Rooted seedlings have the added advantage that they are available year round.

Black and white mangroves have small propagules that must remain on moist substrate for several days in order to germinate and anchor. Hence, they are not practical for installation because of the likelihood they will be lost by water movement (Lewis and Haines, 1981). Crewz and Lewis (1991) suggested that broadcast dispersal of black mangrove propagules might be successful in dense stands of *Spartina* located at appropriate elevations.

Details of handling and transplant techniques for small (0.5-1.5 m) red, white, and black mangrove trees are described by Pulver (1975). Pulver (1975) and Evans (1977) observed that pruning immediately before or after transplanting enhanced recovery and growth, despite some initial defoliation. Pruning had the best effect on white mangrove, *Laguncularia*, which grew 30.6 times faster than unpruned transplants. Pruned *Rhizophora* and *Avicennia* grew 2.0 and 1.6 times faster than unpruned trees. Estevez and Evans (1978) compared mangrove hedges (pruned from above) and trees grazed by cattle (pruned from below) and concluded that thinning the lower canopy caused less reproductive loss than topping. They recommended leaving the top 50% of the tree's final height after cutting in the canopy, not cutting the growing ends of branches, letting mature fruit and propagules fall from trees before pruning, and pruning between February and March in freeze-prone areas and between October and December in other areas. Lewis (1982) reported limited success in moving large mangrove trees, emphasizing that trees must be replanted at the same level in the ground and at the same tidal elevation as their original habitat. Evans (1977) reported that small white mangrove trees could be transplanted during any season, but survival was best for trees transplanted in spring.

Trophic Web Considerations

Teas (1981) recommended accelerating the development of detritus based food webs characteristic of mangrove habitats by adding litter components (leaves and branches) as a source of organic matter after plantings have become established.

Mangroves, whether transplanted or wild, are subject to natural mortality from inter- and intra-specific competition (Pulver, 1975) and predators, including crabs (Lewis, 1989) and boring isopods (Hannan, 1975). Probable mortality rates from competition should be factored into decisions about initial planting densities. Areas known to be infested with the isopod *Sphaeroma* sp. may not provide appropriate site for creation of mangrove wetlands.

Human Interference

Human interference includes trampling, mowing, pruning, digging for bait (e.g., fiddler crabs), vehicular use, dumping, and vandalism. All of these activities can impair the quality of mangrove wetlands. Additionally, alteration of freshwater inputs by ditching, toxic and nutrient runoff, insect spraying, domestic animal damage, and disruption of the activities of mangrove fauna (e.g., nesting, roosting, feeding) through human presence (docks, boating) can disrupt the structure and function of mangrove wetlands. Crewz and Lewis (1991) recommended that sites vulnerable to public access be protected with structures that deter intrusion (signs, barriers such as fences, waterways or vegetative buffer zones). Vegetation buffer zones make sites less obvious and can filter nutrient and pollutant runoff into the swamp (Zedler, 1984). Protective structures include buffers cleared of exotic vegetation (Lewis, 1989). Such buffer zones should be maintained until the regulatory agency responsible for monitoring has deemed the restoration/creation a success.

As an example of the extreme to which human interference may affect mangrove creation or restoration projects, Fehring et al. (1979) described a failed restoration project in Tampa Bay, Florida. The goal of the restoration was to recreate a vegetated shoreline and associated biological communities outside a new seawall. Failure was attributed to a number of factors, including bad faith on the part of the real estate developer, use of an inexperienced contractor, poor construction, and human interference. Residents of property abutting the restoration site apparently deliberately damaged and removed planted mangroves. Teas (1975) noted that human interference was a problem for some mangrove plantings when he monitored in Florida.

3.2.1.2.3.2 Recommended Actions Following Oiling of Mangrove Swamps

Recommended actions following oiling of mangrove habitat are discussed in Section 5.2.1.2. Appropriate response and restoration actions are determined in a hierarchical fashion, depending on whether or not the oil has penetrated the substrate, is adhering to the substrate, is recoverable, the vegetation is contaminated, and vegetative mortality has occurred.

Few long-term studies of the recovery of mangrove forests after oiling have been performed, probably because of the relatively long recovery times involved. Most studies have focused on mangrove trees rather than the complex biological community associated with them (Johnson and Pastorak, 1985). The impact of oil discharges in mangrove habitats is a function of a number of factors including forest location, mangrove species, intensity of oiling, type of oil, and the size of area impacted. In general, sheltered forests are less likely to be cleansed naturally by waves and tides than exposed forests, some mangrove species are more sensitive to oiling than others, and some oils are more toxic than others. For example, light distillates are more toxic than heavier fuel oils. Death and decay of fallen dead trees may increase erosion and further alter the habitat. Recolonization is a function of available seeds or seedlings, particularly if the affected area is large or if currents prevent seeds from settling. Fallen trees moved by waves and tides in overwash forests may impede recolonization by preventing new seeds from surviving (Odum and Johannes, 1975).

Although few restoration cases are well documented, it is clear that cleanup activities in mangrove habitats have the potential to cause greater injury than that inflicted by oiling. There is general agreement that techniques such as steam cleaning, sand blasting, high pressure flushing, and methods involving heavy equipment, including digging of channels, should be avoided when attempting to remove oil from mangrove forests (Johnson and Pastorak, 1985; Levings et al., 1993). Consequently, natural recovery is recommended as the best recovery strategy in exposed mangrove habitats, allowing natural cleansing by waves and tides. A similar recommendation is made for sheltered mangrove forests (Getter et al., 1981). If oil must be removed to avoid recontamination, low pressure flushing may be performed from boats, provided oil has not penetrated the substrate. After a reasonable period of time, if natural recovery is not underway due to a lack of colonizing seeds and propagules, replanting should be considered.

3.2.1.2.3.3 Recovery Times Following Oiling of Mangroves

The time to complete recovery of oiled mangrove habitats has not been measured, although most authors cite time scales of decades ranging from 20 (Johnson and Pastorak, 1985; Burns et al., 1993) to 80 (Johannes, 1974) years. At a minimum, recovery time will equal the time required for trees to reach maturity. Burns et al. (1993) concluded that the toxic effects of oiling will probably persist for at least 20 years in deep mud tropical coastal habitats affected by catastrophic oil discharges.

The results of extensive mortality of mangrove forests in Vietnam during the 1960s and 1970s may be pertinent to estimation of recovery time. The chemical herbicides applied in Vietnam, primarily Agent Orange (normal butyl esters of D,4-D and D,4,5-T in a 1:1 ratio) and Agent White (triisopropanolamine salts of 2,4-D and picolram in a 4:1 ratio), killed mangroves outright rather than simply defoliating them. Tschirley (1969) estimated that a minimum of 20 years was required for recovery of the dominant *Rhizophora-Bruguiera* complex following application of herbicides in Vietnam. Orians and Pfeiffer (1970) and Westing (1971) argued that a longer recovery period was likely because the supply of mangrove seeds to defoliated areas was limited and germination conditions in the defoliated forests were not optimum. Westing (1971) reported that little recolonization, even by opportunistic invaders, had occurred 6 years after spraying.

3.2.1.2.3.4 Monitoring Mangrove Swamps

The importance of efficient monitoring programs following creation and restoration of wetlands was emphasized by Crewz and Lewis (1991), who noted that evidence of the need for monitoring is obvious from the damage observed at many of the sites that they monitored. Damage included slope erosion, encroachment from adjacent construction, debris impacts, and drainage impairments.

Critical elements of an efficient wetlands monitoring program are a time-zero site report, a statistically sound sampling program, and flexibility for timely remedial actions as problems arise (Crewz and Lewis, 1991). These elements are discussed separately below.

<u>Time-zero report</u>: The time-zero report should be prepared immediately following site restoration. It should include descriptions of biotic and abiotic site characteristics, as-built large-scale drawings that document plant locations by species, soil type distributions, slopes and elevations of margins, information on planting dates, etc. Aerial and ground-level photographs of the site should be included. A semi-permanent benchmark should be established and its precise location recorded.

<u>Sampling program</u>: A statistically sound sampling program that employs accepted scientific techniques should be used to measure pertinent site characteristics at each monitoring visit. Standard vegetative variables for trees are percentage cover by species, density, survival, colonization, basal area, diameter at breast height, vegetation height, above ground biomass, leaf area index, and crown spread. Functional variables including rates of primary production and trophic transfer should be measured so that the functional, as well as structural, equivalency of the created wetland can be compared to a reference site (Moy and Levin, 1991; Citron-Molero, 1992). If the site was constructed to provide animal habitat, animal presence at the site must be recorded over at least a 24-hour period.

Crewz and Lewis (1991) recommended that monitoring begin immediately upon site restoration. Following completion of site planting, monitoring should be conducted frequently through the first six months, with quarterly, and eventually biannual, sampling conducted. Written reports and photographs should be submitted to the appropriate regulatory agency at the beginning of the project and immediately as problems are observed. Patterson (1986) described aerial photographic techniques in which different mangrove species assemblages and habitat types were characterized by characteristic spectral signatures, permitting rapid synoptic surveys of mangrove environments. Pre-incident baseline data should be used if available, and unoiled reference sites should be established. The oil content of mangrove substrate should be measured in sediment cores.

Mid-course alterations: Remedial actions may be needed to correct problems if a site is not developing properly. For example, elevations may be inappropriate, flushing or drainage may not be adequate, or plant material may be poor. Timely mid-course alterations may correct these problems and increase the chances that the wetland will mature.

Ideally, oil-impacted mangrove swamps should be monitored over a time period appropriate to document recovery. The timescale of monitoring will be discharge- and location-specific. Ideally, the minimum monitoring time is equivalent to the time to maturity of adult mangroves, generally on the order of more than one decade. As a practical matter, Crewz and Lewis (1991) recommended monitoring for a minimum of five years in mangrove wetlands. Monitoring over this period may be adequate for establishing short-term survival of installed plants, but longer monitoring programs, coupled with mid-course alterations, will improve the likelihood that a site matures and that restoration is successful.

3.2.1.2.3.5 Recommendations for Future Research

Future research needs include development of non-destructive response methods to oiling of mangrove habitat, including bioremediation. The timescales of recovery of functional values, including nutrient pools, biomass production, and trophic transfers need to be better understood.

3.2.2 Freshwater Wetlands

Freshwater wetlands encompass a wide diversity of habitat types in riverine and palustrine environments, including emergent marsh, scrub-shrub wetlands, and forested wetlands. Palustrine environments also include unusual wetland types such as bog and fen habitats, vernal pools, prairie potholes, and kettles. Few studies of the impacts, long-term effects of oil discharges, or recovery and restoration following oiling of freshwater wetland habitats have been published. Those that exist concern almost exclusively riverine habitats and arctic and subarctic tundra habitats. Below, the available literature on oiling of freshwater wetlands is reviewed by habitat type. Studies of freshwater wetland restoration that do not involve oiling (the majority of the literature) are reviewed separately.

3.2.2.1 Riverine Wetlands

3.2.2.1.1 Riverine Emergent Wetlands

3.2.2.1.1.1 Case Studies of Oil Discharges in Riverine Emergent Wetlands

St. Lawrence River Discharge

In June 1976 the NEPCO-140 barge hit shore in the St. Lawrence Seaway shipping lane, discharging 7,310 barrels of No. 6 fuel oil into the Saint Lawrence River. Swift currents in the channel carried 308,000 gallons of oil downstream within a few hours (Alexander et al., 1979). Most of the oil washed into an emergent *Typha* marsh. Immediate post-discharge mortality of fish, frogs, turtles, ducks, geese, herons, and muskrats was reported. Response efforts consisted of removal of the oil and cutting of the oiled vegetation below water level (Alexander et al., 1981).

The impact of the discharge on the cattail marsh was monitored for two years following the discharge. Pre-incident data were not available, but some vegetation had been mapped prior to the discharge. Four sites were monitored: a heavily oiled site, a moderately oiled site, a lightly oiled site, and an unoiled reference site (Alexander et al., 1981). Statistical methods were not described and monitoring appears to have been qualitative. Cattail growth was normal at all sites during the first spring after the discharge. By June, sites that had been oiled and cut exhibited higher growth than unoiled sites. However, no flowering of cattails occurred at these sites by the end of the summer. Two summers after the discharge, normal flowering occurred at all sites. The authors cited subsequent separate studies to the effect that increased growth at oiled sites was due to nutrients in the oil. Complete recovery of *Typha* marsh was estimated to occur two years after oiling and cutting.

Connecticut River Marsh

In January 1972, 3,800 liters of fuel oil were discharged into Mill River, a tributary of the Connecticut River in Northampton, Massachusetts. No response efforts were reported (Burk, 1977).

Burk (1977) surveyed three vegetation zones---high marsh, mid-marsh, and low marsh---for 44 months following the discharge. Line transects with permanent quadrants were established. Percent plant cover, number of species, and species diversity were determined 8 months, 21 months, 32 months, and 44 months after the discharge. Some pre-incident baseline data were available from August 1971, five months prior to the discharge. While basic statistics describing variance were calculated, statistical tests were not performed. Overall, annual plants were most affected one year after the discharge with many species eliminated. Perennials recolonized during the second year after the discharge and annuals reinvaded during the third and fourth years after the discharge. Burk (1977) noted that factors other than oil may have influenced recovery of the marsh, including unusual summer floods, raw sewage released into the marsh during 1974, and introduction of Canada geese in 1974-1975. Recovery time was not estimated.

Little Panoche Creek, California

In September 1974, a pipeline break discharged 31,000 barrels of San Joaquin Valley heavy crude oil into Little Panoche Creek, located near Fresno City, California. The oil saturated vegetation and soil along two miles of the creek. Response activities involved placement of booms to concentrate and divert oil into the creek and away from adjacent pond environments. Four separate impoundment areas to collect the oil were created by excavating an existing ravine. Oil was then vacuumed from the impoundments. Sorbent booms and traps were deployed to trap the remaining oil. Oiled vegetation and soil were removed from the creek edge (Pimentell, 1985).

Restoration efforts involved replacing the creek contours where soil had been removed, creation of additional wetland areas by constructing small berms to divert water flow into adjacent saltgrass flats, and increasing water ponding within the creek in order to promote growth of vegetation and recruitment of fish. Four sections of creek bottom were widened and layered with gravel. The creek banks were replanted with brush. Subsequent monitoring appears to have been qualitative. Quantitative surveys were not reported and statistical analyses were not performed. Vegetation had regrown in the replanted areas and begun to grow in the enhanced areas one year after the discharge. Because of the pond creation effort, there was some loss of marsh area. Fish colonization of the ponded areas was interpreted as evidence that the original water quality was restored. Hence, time to recovery was estimated to be one year (Pimentell, 1985).

3.2.2.1.1.2 Non-oil Restoration Studies of Riverine Emergent Wetlands

Kissimmee River

The Kissimmee River was once a broad, meandering waterway that drained an upper basin consisting of a chain of lakes in south central Florida. Historically, water flowed overland through natural streams near Orlando, through an expansive marshy floodplain, and into Lake Okeechobee, its southern terminus. During summer high water periods, the lake overflowed its southern banks into the Everglades. Although the connection between the lake and the Everglades was intermittent, the habitat that received periodic flooding was continuous (Whitfield, 1986; Berger, 1992).

Channelization of the upper basin of the river system for flood control between 1961 and 1971 transformed a 103 mile long meandering river into a deep 56 mile long canal between Lake Kissimmee and Lake Okeechobee. Channelization drained 34,000 acres of Kissimmee floodplain wetlands and converted 13,000 acres into impounded wetlands. Much of the post-channelization wetland acreage differed qualitatively from the original wetlands due to more static water levels in the channelized system. Channelization caused profound changes in the hydrology of the area in terms of hydroperiod, amount of flow, rates of flow, and distribution of flow. Water quality in the river, the Everglades, and Lake Okeechobee deteriorated (McCaffrey, 1985; Berger, 1992).

The Save-Our-Everglades program, a large, publicly funded effort established in 1983, included rehabilitation of the Kissimmee River by restoring natural flow. A demonstration project to assess the ecological effects of reflooding was undertaken in 1984 (Berger, 1992). Three notched wiers were constructed in the channelized river in order to reflood 1,300 acres in remnant sections of the river channel. Remnant oxbows along 12 miles of the canal were reflooded in August 1985. Effects of reflooding on floodplain vegetation, fish, secondary production, and benthic invertebrates were monitored for five years. By January 1987, water had begun to drown native, but out-of place, wax myrtle and oak (Glass, 1987).

The reproductive potential and seedbank of many wetland plants were preserved even after two decades of drainage, and following reflooding, wetland vegetation and wildlife recolonized rapidly. Because the extent and depth of flooding and drying were not comparable to prechannelization conditions in many parts of the reflooded area, the demonstration project was only partially successful as an ecosystem restoration. However, the project was a significant success in showing that a riverine-floodplain ecosystem could be restored (Berger, 1992).

As of 1992, a final restoration plan for the area was under design. A 10-15 year effort was envisioned, with the following broad guidelines:

- The restoration should use the natural free energy of the river system, rather than that of an impounded, managed system;

- The natural ecological functions of the river should be restored;

- The physical, chemical, and biological integrity of the river system should be restored and maintained; and

- Lost environmental values should be restored.

To evaluate success in achieving these goals, highly specific criteria were established with respect to flow duration and variability, flow velocities, stage-discharge relationships, stage recession rates, and inundation frequencies (Berger, 1992). When completed, restoration of the Kissimmee River will be the largest wetlands restoration project undertaken in the United States. A period of years to decades will be required to evaluate its success.

Cypress Creek, Florida

Devroy and Hanners (1988) described the restoration of a channelized 243 ha swamp in Florida. Cypress Creek, which drains a 30.3 km watershed, was channelized for flood control in 1962. In August 1986, the southern part of the system was dammed in order to restore natural floodwater storage capacity and prevent downstream flooding. Three transects and six 1 m^2 vegetation plots were monitored for water level, plant species composition, and plant areal coverage at five quarterly intervals after dam construction. Results were compared to baseline data from the same area obtained in 1984. One year after damming, shallow groundwater levels had increased significantly and hydroperiod was restored. Vegetation changed in some areas, but the overall results were ambiguous. The authors concluded that long-term monitoring was necessary to evaluate revegetation success. Such long-term monitoring was not reported and success was not evaluated.

Allegheny River, New York

Pierce et al. (1981) described a pilot project for wetland reclamation in the Allegheny River floodplain in Cattanaugas County, New York. The project to create 1.8 ha of marsh began in 1981. Native, locally-collected emergent macrophytes were planted in 180-100m^2 subplots during October and November, after the onset of frost, but before freezing conditions occurred. Controlled experiments to examine the effects of water level, substrate, and fertilizer were performed on 16 plant species and compared to other sites. Planting materials tested included seeds, rhizomes, cores, mulch, natural generation, and combinations of these materials. Muskrat were excluded by fencing, trapping and shooting. Damage by deer and waterfowl was countered by replanting. The success of the various planting methods was not reported nor was the success of the restoration effort evaluated.

Fifteen Mile Creek, Oregon

Kentula (1986) described restoration of streamside vegetation along Fifteen Mile Creek, located south of the Dalles River, Oregon. Vegetated habitat along the creek had deteriorated due to grazing, agriculture, timber harvest, and withdrawal of water for irrigation. Restoration began along an 8-10 mile length of stream in 1974 with planting of wetland species. The plantings were protected by unspecified means and streamside vegetation was said to have recovered within four years after planting. A detailed evaluation of success of the restoration effort was not performed.

3.2.2.1.2 Riverine Scrub-Shrub Wetland

3.2.2.1.2.1 Case Studies of Oil Discharges in Riverine Scrub-Shrub Wetlands

Santa Ana River Drainage Discharge

The single study of oil impact on riverine scrub-shrub wetland is of willow thickets in the Santa Ana River drainage. In January 1983 5,000-7,000 gallons of crude oil discharged from an abandoned well into the Prado Flood Control Basin in Riverside County, California. The impacted area was a forested wetland supporting a variety of wildlife, including migratory waterfowl. The oil washed into dense willow thickets near the center of the basin and into two duck club ponds. Initial cleanup efforts involved deployment of containment and sorbent booms, and straw and wood chips to concentrate the oil. Oil and oil-soaked debris were removed manually using small recreational-type aluminum boats that provided the only access to the contaminated areas. Oil and debris were recovered using screen-covered rakes and pitchforks. The densest thickets were flushed with water sprayed from gas-powered pumps (Kemerer et al., 1985).

Pre-incident baseline data were not available and recovery was not documented quantitatively. Statistical analyses were not performed. Kemerer et al. (1985) reported that no effects of the discharge were visible six months later during the dry season. Time to recovery was not estimated.

3.2.2.1.2.2 Non-oil Restoration Studies of Riverine Scrub-Shrub Wetlands

Jarman et al. (1991) compared six created wetland sites in Massachusetts, which included scrub-shrub habitat. Success was defined according to Massachusetts state regulations as establishment of 75% cover within two growing seasons. Functional equivalency of the created wetlands was assumed, but functional analyses were not performed. Vegetation establishment was successful at the six sites, but species composition differed from that of adjacent wetlands. Survival rates varied with species and transplanting technique. Survival of shrubs transplanted from adjacent habitats was generally poor, but survivorship of nursery grown shrubs was high. Wetland soil conditions had begun to develop within the two years during which monitoring was performed. The use of organic soils transferred from lost wetlands expedited establishment of indigenous wetland vegetation, but establishment of the herbaceous community alone fell far short of the project's long-term goal of in-kind replacement.

Willard et al. (1990) listed a number of mitigation projects in the midwestern riverine scrub-shrub habitats, but the level of detail given is insufficient to evaluate success.

3.2.2.1.3 Riverine Forested Wetlands

No studies of oil impacts or restoration following oiling of riverine forested wetlands were located. Studies involving restoration of bottomland hardwood forests in the lower Mississippi valley and riparian habitat in the arid southwest are reviewed below.

Bottomland hardwood forests are characterized by rapid growth rates and high production, reflecting the influence of rich soils, long growing season, and high rainfall. Species diversity is moderate to low, restricted to flood-tolerant forms. Newling (1990) and Sharitz (1992) described several large-scale restoration efforts, including forested wetlands, in the lower Mississippi River valley. Most of these restoration efforts focused on reestablishment of forest species for timber or wildlife habitat value. The emphasis in such efforts has been to establish forest canopies of selected species, particularly oaks and other heavy-seeded trees with limited dispersal. Trees of other species and undergrowth plants are generally ignored, or expected to become established naturally, with the overriding concern to produce tree canopy over large tracts of land (Clewell and Lea, 1989). Most such bottomland forest restorations do not attempt to restore original physical and hydrologic conditions. Actual site restoration, including recovery of original hydrologic conditions is uncommon, and success is typically measured on the basis of early establishment of desirable woody tree species. Bottomland forest replacement requires decades and techniques are not well developed. Functional equivalency is usually not addressed (Clewell and Lea, 1989).

After soybean prices fell in the early 1970s, large tracts of agricultural land in the lower Mississippi valley were abandoned. By 1987, the U.S. Fish and Wildlife Service had completed plantings to restore bottomland hardwood habitats on parcels ranging from 1/2 acre to 400 ha in a number of areas in southeastern Mississippi. The Mississippi Department of Wildlife Conservation had established a 10-year project to restore 400 acres of the Malmaison Wildlife Management Area and the Louisiana Department of Wildlife and Fisheries had begun restoring 1600 ha at two management areas. Newling (1990) estimated that 9,000 ha of wetlands had been restored or enhanced in six southeast states by 1989.

Newling (1990) noted that hardmast (i.e., large tree) production does not begin in these habitats for 25-30 years and most stands do not mature for 40-60 years. It takes even more time until such areas resemble old growth forests. This timeline is short, however, compared to recovery times in other regions. Sharitz (1992) noted that smaller-scale areas have a greater possibility of functional recovery than larger areas because it is more feasible to restore the original hydrologic regime. Nevertheless, the goal of duplicating an original forest stand in terms of species composition, age, structure, and function can only be approximated, at best. Natural forests are dynamic systems in constant flux. Further, land use activities may have modified soil and hydrologic conditions such that duplication of the original hardwood forest is not possible.

Carothers et al. (1990) reviewed restoration of riparian habitats in the arid southwest. In the southwest, natural watercourses have been so impacted by man and are so controlled by dams that it is rarely possible to create conditions suitable for revegetation. Thus, most such efforts involve planting trees and generally do not involve creating conditions for natural revegetation. Riparian plant species such as cottonwood that depend on floods for successful seed germination, no longer reproduce naturally in these areas. Restoration of natural flow conditions is unlikely to occur because of water demands and residential and agricultural uses of floodplains. Because of changes in hydrological conditions, most restored riparian forests have not reproduced and are unlikely to do so. Because their longevity is equivalent to the lifespan of the individual trees, perpetuation of such restored forests requires maintenance programs of periodic replantings.

The oldest revegetation project along the lower Colorado River contains trees planted approximately 14 years ago (Anderson and Ohmart, 1979; Manci, 1989; Carothers et al., 1990). Salt cedar, an opportunistic species, was cleared from the site and the site was levelled. Two thousand willows and cottonwoods were planted in augured holes in January 1979 and the trees were irrigated for several years until their roots reached the water table. Restoration appeared to be successful. However, Carothers et al. (1990) noted that a recent inspection of the site revealed that all planted willows and many planted cottonwoods were moribund.

3.2.2.2 Palustrine Wetlands

3.2.2.2.1 Palustrine Emergent Wetlands

No studies of oil discharges in palustrine emergent wetlands were located. Restoration studies that do not involve oiling of marsh, reservoir shoreline, prairie pothole and vernal pool habitats are reviewed below.

3.2.2.2.1.1 Non-oil Restoration Studies of Palustrine Emergent Wetlands

Corkscrew Swamp, Florida

Carlson (1982) reported preliminary results following restoration of farmed freshwater marshes in the Corkscrew Swamp Sanctuary in Collier County, Florida. The 180 ha wetland area had been modified for vegetable farming during the 1950s. Farmed areas were surrounded by earthen dikes with adjacent parallel ditches and contained interior dike and ditch networks for water control. The resulting ecological impacts ranged from localized, but drastic, changes in ground elevation, hydroperiod, and plant communities to broadscale alterations in surface water flows.

Restoration efforts began in the spring of 1981. Dike material was removed to adjacent ditches with earth moving equipment in order to restore the profile of 60 ha of farmed marsh. Precise releveling was not possible because of differential accumulation of organic material on the created dikes and ditches. Monitoring consisted of an extensive photographic record of the site before, during, and after restoration. Vegetation transects in restored and control areas were censused at unspecified intervals for species composition, percent cover, height and biomass. Vegetative recovery on the restored ditches was said to be complete after one growing season, while recovery on the dikes was minimal. Because of the short duration of monitoring, the success of the effort as a marsh restoration cannot be evaluated.

Wisconsin Marshes

Owen et al. (1989) compared natural and restored marshes at two sites in Wisconsin. One restoration site, located in Green Bay, was centered around a pond with slopes too steep for development of wetland plants. After 10 years, the site was covered primarily with upland weeds or opportunistic wetland plants. In contrast, adjacent natural wetlands were characterized by soft, peaty soil, gradual slopes, few ponds, and abundant wetland vegetation. Hence, the restoration was deemed unsuccessful. The second restoration site, located in Madison, was created by transferring salvaged marsh surface from an area destroyed by highway construction. This site had more gradual slopes, more organic substrate, and a more natural variety of wetland vegetation after 2-3 years. Problems with the restoration included imprecise grading, which resulted in an altered hydrologic regime and some invasion by weed species. The success of this site as a restoration effort was not evaluated.

Colorado Cattail Marsh

Buckner and Wheeler (1990) described construction of a 5 ha cattail marsh in eastern Colorado in 1986. The site was chosen on the basis of suitable topography and soil characteristics. It had been a wetland until the 1940s. Three 46 cm high spreader berms were installed to encourage even spreading of water. Two hectares were planted using "live topsoil" removed from a nearby marsh doomed by highway construction and 3 ha were seeded one year later with cattail seed collected locally the previous fall. After one year, plant material in the topsoiled area germinated slowly, but steadily, in May and June, resulting in 48% cover by September (30% of the area was open water). Seeded areas germinated by mid-May, resulting in 77% cover by September (8% of the area was open water). Because of the short duration of monitoring, the success of the project as marsh restoration cannot be evaluated.

Hole-in-the-Donut, Florida

Doran et al. (1990) and Bacchus (1991) described restoration of former marsh wetlands at Hole-in-the-Donut in Everglades National Park, Florida. Four thousand hectares had been farmed intensely for several decades using crude mechanical soil preparation methods. Rock-plowing was developed in the early 1950s in order to crush the natural limestone rock and apply fertilizer to create better soil for crops. Consequently, substrate in the area changed from low nutrient, anaerobic conditions to higher nutrient, aerobic conditions accompanied by invasion of an opportunistic exotic plant species, Brazilian pepper (*Schinus terebinthifolius)*. Control of Brazilian pepper was attempted by a number of techniques including planting, mowing, burning, bulldozing, and substrate removal. Only substrate removal was effective in increasing hydroperiod and altering the successional pattern in favor of natural revegetation. Substrate was removed from 24.3 ha in 1989, and hydrology, microbiology, nutrients, and vegetation were monitored. Preliminary results suggested that hydrological and substrate conditions in the restored site favored succession toward native marsh vegetation. As of February 1990, 56% of plant species were wetland forms. However, the success of the project as a restoration effort was difficult to evaluate because adjacent, untreated rock-plowed land also returned to wetland after being abandoned by agriculture.

Reservoir Shoreline

Development of wetland plant communities on the shore of a new reservoir was described by Hooker and Westbury (1991). The reservoir was created by impounding a creek near the Savannah River, South Carolina. A large effort was undertaken in which 4,270 linear meters of shoreline were planted with approximately 100,000 plants of 51 species. Transplants from adjacent ponds, creek branches, nursery stock and seeds were planted in five colonization zones based on slope and distance from shore. After four years, littoral plant cover was highly variable, ranging from 4.4-73.6%. Cover in unplanted control areas was low overall, apparently dependent on shoreline and substrate characteristics. In planted areas, wetland fringe species diversity was comparable to that observed at two older cooling ponds in the same area. Success of the project as a restoration effort was not evaluated.

3.2.2.2.1.2 Prairie Potholes

The prairie pothole geologic region encompasses about 192 million acres in Alberta, Saskatchewan, Manitoba, Montana, North Dakota, South Dakota, Minnesota, and Iowa. The region is characterized by relatively flat glacial topography with poorly defined natural drainage and millions of potholes distributed across the landscape. The area was heavily altered for farming beginning in the mid-19th century with seasonal and perennial inundation of potholes eliminated by installation of drain tiles and outlet ditches. Hay (1992) estimated that wetlands in the area were reduced by 50% in the period between the 1870s and the 1970s.

Hay (1992) reviewed 18 prairie pothole restoration projects in Meeker and Rice Counties, Minnesota. All of the sites were restored by the U.S. Fish and Wildlife Service on private agricultural lands. The primary goal was restoration of waterfowl habitat. Secondary goals included flood control and water quality improvement. Restoration involved simple changes to drainage systems. For example, on one property, agricultural drainage structures in and around 10 farmed potholes ranging in size from 0.2-10 acres were removed, blocked, or altered in order to emulate pre-settlement conditions. The tiles draining the potholes were blocked. Drainage ditches were blocked by small earth fills or dikes and each dike incorporated a small spillway. On another property, drainage structures were modified for potholes of 0.7 and 1.5 acres. Earthen dikes were used to block surface drainage and tiles were removed to prevent subsurface drainage. No plant materials were introduced into the potholes being restored and a limited number of plant species, both warm- and cool-season grasses, were planted in the buffer areas around the restored potholes. Formal monitoring was not done and success with respect to site-specific and regional objectives is unknown. Hay (1992) noted that plant diversity in the restored potholes was extremely low and consequently wildlife habitat value was low. Nevertheless, the projects were viewed as successful because the U.S. Fish and Wildlife Service responded to individual, local interests in restoration with the result that potholes were taken out of agricultural use and returned to their natural functions of water storage, nutrient cycling, and wildlife habitat.

3.2.2.2.1.3 Vernal Pools

Vernal pools are endangered wetland habitats that flood annually during winter and support a unique biota. They are found in areas with Mediterranean climate. Pritchett (1990) described the creation and monitoring of vernal pools at Santa Barbara, California. Six pools were created by excavating shallow depressions in clay soil. Three were inoculated with seed bank obtained from local vernal pools, three were not inoculated. The created pools were monitored for one year and compared to adjacent natural pools. After one year, the duration of flooding was longer and more variable in the created pools than in the natural pools. More native plants occurred in the inoculated created pools than in the uninoculated created pools and two annual plant species endemic to vernal pools were more abundant in the inoculated pools than in the natural pools. The success of the project as a restoration effort was not evaluated.

3.2.2.2.2 Palustrine Scrub-Shrub Wetlands

No studies of restoration or creation of palustrine scrub-shrub wetlands were located.

3.2.2.2.3 Palustrine Forested Wetlands

3.2.2.2.3.1 Case Studies of Oil Discharges in Palustrine Forested Wetlands

Baca et al. (1985) cited an unpublished example of a Louisiana cypress swamp in which 30,000 barrels of crude oil were released as the result of a well blow-out in January 1983. Statistical analyses were not reported. Comparisons of control and affected sites one year after the discharge revealed that oil effects on vegetation were species-specific. Areas with high shading by mature trees had little or no understory and few effects of the oil were observed on the dominant woody vegetation. Perennial plants were returning to the sunlit areas. In contrast, oiled areas formerly covered with floating vascular vegetation were devoid of any vegetation. Similar effects were noted in a freshwater swamp discharge in Nigeria. Recovery times were not estimated.

3.2.2.2.3.2 Non-oil Restoration Studies of Palustrine Forested Wetlands

Weston and Brice (1991) described a 3-acre hardwood swamp restoration project in St. Petersburg, Florida. Restoration involved removal of 2 acres of Brazilian pepper, an exotic opportunistic species, treatment of the area with herbicide to inhibit pepper regrowth, and dredging 0.5 acres to create a pond. Approximately 750 native trees, shrubs and herbaceous plants were installed. Planting was completed between February and July 1990. After one year of a three year monitoring program, minimal regeneration of Brazilian pepper had occurred. Survival of planted species was variable: 20% for shrubs, 81% for American elm, 97% for pond cypress, and 100% for aquatic plants in the pond. Growth over the year ranged from 7-71% for the various planted species, and natural colonization was also occurring. Because of the short duration of monitoring, success of the project as a restoration effort was not evaluated.

3.2.2.2.4 Bogs and Fens

The only studies of oil impacts on bog and fen habitats are of arctic tundra and taiga vegetation, i.e., non-forested and forested wetland habitats located in areas characterized by permafrost. Case histories are reviewed in chronological order below.

3.2.2.2.4.1 Arctic and Subarctic Bogs and Fens: Alaska Pipeline Discharges

Hunt et al. (1973) examined four discharge sites along the Haines to Fairbanks pipeline in Alaska during the summers of 1971 and 1972, 4-15 years after the discharges occurred. Sites were chose to reflect a range in terrain and climatic conditions. The discharges are reviewed separately below.

Milepost 1.9 Jet Fuel Discharge

A discharge of JP-4 jet fuel occurred in 1968 in a moist coastal region near Haines, Alaska. The site was located at 122 m elevation with a 15% west-facing slope. All vegetation in contact with the fuel was killed. When Hunt et al. (1973) visited the discharge site in 1972, 4 years after the discharge, fuel was still present below the soil surface. Observations were qualitative, and statistical analyses were not performed. The presence of a luxuriant undergrowth of herbs and shrubs was explained as the result of rain leaching the discharged jet fuel from the uppermost soil layers, allowing the vegetation to regrow. The time to recovery was not estimated.

Milepost 119 Diesel Discharge

A discharge of diesel oil occurred near Lake Dezadeash in the Yukon territories, Canada in 1968. The site elevation was 730m with a 20% east-facing slope. Observations were qualitative and statistical analyses were not performed. The fuel permeated the downslope soil, contaminating areas of different vegetation, including a stand of willow and alder and a stand of intermediate aged white spruce. Both stands had associated understories of mosses and lichens. All willows, except those located on high spots, were killed, as were all white spruce in the fuel's flow path. When Hunt et al. (1973) visited the site four years after the discharge, there was little recovery, even by opportunistic species such as fireweed. The time to recovery was not estimated.

Milepost 197.1 Jet Fuel Discharge

A discharge of JP-4 jet fuel occurred near Kluane Lake, Yukon Territories, Canada in 1956. The discharge site was located in permafrost terrain at 800 m elevation with a 25% slope facing northeast. Observations were qualitative and statistical analyses were not performed. The prespill vegetation consisted of an intermediate-aged stand of white spruce with an associated ground cover of mosses and lichens. Most of the vegetation was killed by the discharge. When Hunt et al. (1973) visited the site 15 years after the discharge, new, small willows and occasional shrubs had regrown. They noted an increase in permafrost depth where fuel had killed the vegetation. The time to recovery was not estimated.

Milepost 207.6 Jet Fuel Discharge

A discharge of JP-4 jet fuel occurred in 1956 at a site located at 790 m elevation with a 50% slope facing northeast. Observations were qualitative and statistical analyses were not performed. Prespill vegetation consisted of a white spruce stand with associated understory. All vegetation was killed by the discharge. When Hunt et al. (1973) visited the site 15 years after the discharge, little recolonization had occurred. Permafrost depth had increased slightly, but not significantly, due to the fuel. Time to recovery was not estimated.

From their visits to discharge sites along the Haines to Fairbanks pipeline, Hunt et al. (1973) concluded that refined fuels are extremely toxic to subarctic vegetation. Revegetation appeared to be controlled by the amount of moisture available to leach oil and enhance plant growth. Hence, vegetation at the moist coastal site near Haines had recovered five years after the discharge. At interior sites with less rainfall, most revegetation occurred in drainage swales where water flow leached oil. Discharges in permafrost areas with thick organic mats did not cause an increase in permafrost degradation. Some increase in thaw was observed, but the organic mats remained intact. Hunt et al. (1973) noted that mechanical cleanup methods were more likely to cause severe permafrost damage than petroleum discharges alone.

3.2.2.2.4.2 Arctic and Subarctic Bogs and Fens: Experimental Studies

Experimental studies of tundra and taiga habitats have been performed to monitor oil discharge effects on vegetation, compare revegetation techniques, and evaluate the enhancement of microbial degradation. These topics are reviewed separately below.

Oil Effects on Vegetation

Wein and Bliss (1973) studied the effects of experimental oilings on five different arctic plant community types in northwestern Canada. Plant communities differed with respect to species, soil, active layer depth, moisture, and microtopography. All were underlain by permafrost, with a biotic gradient ranging from a tree-covered area at Inuvik, located 115 km from the arctic coast, to tundra at Toktoyuktuk, located on the coast. In a factorial design experiment, light gravity sweet crude oil was applied at various doses during three different seasons. Spring and winter doses were 0, 0.25, 0.5, and 1.0 cm; summer doses were 0, 0.4, 0.75, and 1.5 cm. The maximum spring and winter doses were equivalent to 1,300 barrels per acre and the maximum summer dose was equivalent to 1,950 barrels per acre.

All actively growing plant tissue was destroyed. Plant recovery from latent buds on dwarf shrub species, especially *Salix* and *Betula*, was more rapid than for sedges. Lichens did not recover, and only one moss, *Polytrichus junipernum* exhibited any regrowth. Injury was greatest following summer applications, because the oil penetrated deeper into the soil. The extra energy absorbed on the contaminated plots was dissipated as latent heat of evaporation in spring and as sensible heat later in summer, rather than increasing active layer depth. Because total plant recovery was 20-55% on the treated plots after one full growing season, Wein and Bliss (1973) concluded that contaminated areas should be left undisturbed if possible.

Hutchinson and Freedman (1975) studied the effects of experimental summer and winter crude oil discharges on tundra and taiga vegetation at 6 sites in the Northwest Territories, Canada. The taiga study site was a black spruce association located near Norman Wells, Northwest Territories. Part of the site had been burned 30 years prior to the study. The tundra site was located near Toktoyaktuk, Northwest Territories, and included poorly drained and well-drained subhabitats. Permafrost depth exceeded 200 feet at both sites. Norman Wells crude oil was applied by even surface spraying and as high intensity point discharges. Doses were 9 liters m^2 in sprayed areas and one single point 50 barrel discharge. Baseline data were collected from pre-incident surveys. The study areas were monitored for three years following application of oil. Measurements included plant pigments (i.e., chlorophyll a, chlorophyll b, and carotenoids), physiological rates including transpiration and evapotranspiration, light, and soil heat flux.

Oil effects were evident at both tundra and taiga sites within 48 hours of oil application. All surface discharges had a devastating effect on above-ground vegetation, but plant species differed markedly in their ability to survive and recover. Lichens, mosses, and liverworts were killed outright and did not recover during the three years of the study. Some woody and dwarf shrubs were able to produce new shoots within a few weeks of initial defoliation. Reduced production of storage material resulted in increases in plant losses by winter-killing. Plants with thick, waxy cutiles exhibited the least initial injury, but died later. Regardless of discharge season, flowering and reproduction were severely reduced, even during the third summer after oiling. The permafrost was not significantly affected despite changes in energy budgets.

Overall, injury was greater in the exposed taiga sites than at tundra sites. Taiga species with deep or substantial below-ground storage organs were able to revegetate and recolonize. Tundra vegetation was better able to survive discharge effects and regenerate, despite losses of lichens and mosses. Recovery of these sites was attributed to the presence of several key species. Winter discharges had less effect than summer discharges in both tundra and taiga habitats due to the absence of actively growing foliage at the time of the discharge and to weathering of toxic oil components. Point discharges caused less injury than uniform spraying because the discharged oil was absorbed rapidly into the soil and then flowed beneath the surface. As long as a few inches of surface soil was clear of oil, vegetation was able to survive (Hutchinson and Freedman, 1975).

Revegetation Techniques

Brendel (1985) performed experimental studies to compare possible revegetation techniques following a crude oil discharge in January 1981 south of Prudhoe Bay, Alaska. The oil dose was ~37 liters m^{-2}, with oil content of the soil ranging from 60,000-275,000 ppm. Revegetation experiments were done in 1982, 1983 and 1984, one to three years following the discharge. The following techniques were compared: cover with clean material; remove contaminated material and cover with clean material; seed and fertilize; till, seed and fertilize; apply oil degrading bacteria; and no treatment. Parameters monitored were oil content of the soil, grass species survival and yield, and grass growth. Survival and yield of grass was best in areas in which heavy doses (1,000 lb $acre^{-1}$ = 112 gm^{-2}) of nitrogen-phosphorus fertilizer were applied in combination with soil tilling. Grass yield in control areas averaged 0.74 g m^{-2}. In contrast, grass yields in fertilized and tilled areas averaged 6.6 g m^{-2}. After one year, soil oil content was reduced by ~20% in fertilized, tilled treatments.

Bioremediation

Hunt et al. (1973) performed a field experiment to evaluate the effects of enhanced microbial degradation of oil on revegetation. Thirty-six 2 x 1.5 m plots were defined at the site of a 1956 gasoline discharge along the Haines to Fairbanks pipeline. Three replicates each of 12 combinations of phosphorus and nitrogen additions with added grass seed were done. Although most of the grass seed was eaten by birds, considerable recovery occurred by colonization of natural volunteer species after one year. Microbiological activity increased in all fertilizer treatments, so that treatment with higher nutrient doses was of little benefit. Time to complete recovery was not estimated.

3.2.2.2.4.3 Non-arctic Bogs and Fens

No studies of the effects of oiling on non-arctic bogs and fens were located. The basic biology and ecology of most bog ecosystems, including shrub bogs, pocosins, and *Sphagnum* bogs is not well known (Sharitz and Gibbons, 1984; Damman and French, 1987). These environments can be highly impacted by agriculture, drainage, and peat mining. Lowering of the water table as a result of such activities is a major problem (Sharitz and Gibbons (1982).

No studies of restoration of non-arctic bog and fen environments were located. However, Stoltzfus and Munro (1990) reported the results of an experimental study comparing substrate types and transplant methods in constructed *Sphagnum* mesocosms during a five year study. Five cm and 15 cm clumps of *Sphagnum* grew better than *Sphagnum* spread loosely over the surface. The latter were more susceptible to desiccation and disturbance. Surface area coverage increased from 25% to 100% in two years. Sawdust was a suitable medium for *Sphagnum* growth, particularly in combination with water flow through a woodchip underlayer below the sawdust layer.

Damman and French (1987) reviewed studies of the recovery of peat bogs in the glaciated northeast following disturbance by fire. In general, dwarf shrubs recover easily by means of underground rhizomes and regain their original cover within three to four years. However, low *Sphagnum* cover following burning persists for decades, allowing opportunistic lichen species, which are found only in burned bogs, to colonize.

Damman and French (1987) noted that bog environments are extremely delicate. Bog water is nutrient deficient. Any nutrient enrichment, for example from sewage, destroys the vegetation. Oil discharges in such environments might exert similar effects.

3.2.2.3 Freshwater Wetlands Recovery and Restoration: Summary and Conclusions

3.2.2.3.1 Recommended Actions

Because of the paucity of published studies on oil discharges in freshwater wetland habitats and the diversity of habitat types, it is difficult to generalize about oil impacts, effects of response activities, restoration actions, and recovery times. Riverine habitats can be separated from other freshwater environments on the basis of flow regime, which allows the possibility of self-cleansing following oiling.

3.2.2.3.1.1 Cleanup Actions Following Oiling of Freshwater Emergent, Scrub-Shrub, and Forested Wetlands

There are few published examples of cleanup of freshwater marshes, scrub-shrub wetlands, and forested wetlands following oiling. Common sense dictates the same approach as in saltmarsh and mangrove habitats, with a high priority given to avoiding disturbance of the substrate by trampling and the use of heavy equipment. All methods applicable in marine marsh and forested environments should apply as appropriate, including low-pressure flushing, use of sorbent booms, etc., to remove oil.

Because insufficient information is available regarding oiling of prairie potholes, non-arctic bogs and fens, and vernal pools, specific recommendations concerning cleanup and restoration of these habitats following oiling are not made. It should be noted that these habitats are unlikely to experience the massive oilings that occur in marine and riverine environments subject to ship traffic unless they are located in oil-producing terrain. For this reason, the only recommended actions are natural recovery and bioremediation (Figure 5.13). The latter remains untested, but may be helpful in accelerating degradation of oil contamination. Arctic and subarctic bogs and fens are discussed separately in section 3.2.2.3.1.4.1.

3.2.2.3.1.2 Restoration Actions Following Oiling of Freshwater Emergent, Scrub-Shrub, and Forested Wetlands

As with saltmarsh and mangrove habitats, natural recovery of freshwater wetlands should be the primary course of action, whenever possible. If restoration is deemed necessary, the same general factors that apply to oiled marine wetlands apply. The following guidelines are compiled from a number of sources. Restoration recommendations for marsh environments are from Ross et al. (1985), Allen and Klimas (1986), Erwin (1989), Gryseels (1989), and Bacchus (1991). Data concerning appropriate marsh plants and planting methods are provided by Wentz et al. (1974), Landin (1978) and Hammer (1982). Restoration recommendations for scrub-shrub wetlands were not located, but should be similar to those for forested wetlands and riperian habitats. Restoration recommendations for hardwood forested wetlands are from Baca and Ballou (1989), Bacchus (1989), Clewell and Lea (1989), Landerman (1989), Denton (1990), Ford and Neely (1990), and Newling (1990). Restoration recommendations for riparian woody vegetation are from Kentula (1986), Gore and Bryant (1989), Mancini (1989), Carothers et al. (1990), Willard et al. (1990), and Rieger (1992). Because of their general similarity, recommendations for marsh, forested and riparian habitats are combined below.

Physical and Chemical Factors Influencing Restoration of Freshwater Wetlands

Hydrology: Hydrology is agreed to be the most critical factor affecting success of all types of restored and created wetlands. Sharitz (1992) noted that watershed effects must also be considered, since activities in upstream portions of a watershed may affect downstream areas. Willard et al. (1990) emphasized the importance of relying on the natural hydrography of restoration sites, i.e., the locations of former natural wetlands are most suitable for wetland restoration or creation efforts.

In arid riparian regions, depth to the water table is a critical factor that must be considered in restoration efforts. Vegetation roots must reach the water table in order to be free of irrigation requirements (Carothers et al. (1990).

Substrate: In general, substrate must be suitable for plant root penetration, and its water capacity and chemical properties must support plant growth. There must be an adequate volume of soil for rooting and exploitation of moisture and nutrients by plants. Factors affecting rooting volume include depth to the wet season water table, soil bulk density, and compaction (Clewell and Lea, 1989). In arid riparian habitats, soil condition and texture are critical. Heavy clay content inhibits revegetation and prevents irrigation water and rainfall from reaching the water table. Because high soil salinity reduces the survival of many species, areas with high salinity soil should be avoided as restoration sites. Areas with moderate salinity soils may require leaching before planting and during irrigation (Carothers et al., 1990). Willard et al. (1990) recommended liming as a means of altering soil pH, if necessary, prior to planting in midwestern riparian habitats.

Site elevation and topography: Site elevation and topography must be appropriate for the wetland community being restored or rehabilitated. In site preparation, excessive use of heavy equipment can lead to creation of a solid, unsuitable soil layer that restricts root penetration. In many habitats, tillage enhances growth of vegetation. In some bottomland hardwood habitats, site preparation does not appear to be critical, although plowing seems to enhance establishment of desirable seedlings by reducing competition from weeds (Newling, 1990). Topsoiling or mulching with topsoil from a donor forest or marsh may inhibit competition from undergrowth during initial growth of new vegetation (Clewell and Lea (1989). Because new stems appear to be suppressed by existing stems, removal of competing vegetation can increase emergence and growth rates of new willow shoots in arid riparian habitats (Manci, 1989). Site design should include appropriate erosion control, if necessary.

Planting Rationale and Techniques

In all wetland habitats, plants should be selected on the basis of their site-specific suitability in terms of growth rate, drought resistance, and tolerance to the chemical characteristics of the substrate.

Marshes: Natural seedbanks removed and relocated from adjacent or similar areas may be used for revegetation. Seedbanks have the advantage of being potential sources of multiple species. However, Willard et al. (1990) cautioned that seed germination patterns and eventual vegetation distributions may be unpredictable and patchy and that problems with weed invasion may occur. Baccus (1989) described an example of the latter problem at a restoration site in Florida, and recommended careful inspection of seedbank donor sites prior to transfer of material. An additional caution is that donor areas should not be denuded or significantly affected (except in areas already slated for destruction). Wentz et al. (1974) compiled an annotated bibliography of freshwater marsh plants and plant establishment techniques. Landin (1978) provided an annotated bibliography of wetland plants growing on dredged material throughout the United States.

Forested wetlands: Potential plant material includes natural seedbanks, seeds, bare root seedlings, containerized seedlings, stem cuttings, and transplanted saplings or larger trees (Clewell and Lea, 1989). Because forest soils contain seeds which can remain viable for many years, seeds from earlier successional stages are usually part of the seed bank, while soils from late successional ecosystems contain fewer seeds. Thus, with a proper seedbank, it should be possible to use forest soils to create new wetland forests (Ford and Neely, 1990). However, this approach is apparently untested. Bare root seedlings survive and grow well in moist substrates. Clewell and Lea (1989) recommended that they be grown from local sources for revegetation projects and planted when hardened or fully dormant. Containerized seedlings are appropriate for sites too harsh for the survival of bare-root seedlings and can be planted later in the growing season than bare-root seedlings. Stem cuttings can be successfully grown from tree species such as willow, sycamore, green ash and sweet gum. Saplings are expensive to transplant and the risk of mortality is high, even if properly balled, bagged and pruned. Clewell and Lea (1989) noted the utility of nurse crops in assisting the establishment of planted trees by stabilizing the substrate and providing shade. For example, they suggested the use of cottonwoods or willows as nurse species for bottomland hardwoods.

Sharitz (1992) cited a publication by Allen and Kennedy (1989) on general reforestation techniques for landowners. The publication provides guidance concerning planting techniques, including seed sources, seed storage, site preparation, planting depth and spacing, commercial sources of seedlings, and information concerning the flood tolerances of bottomland forest species. Planting strategies for heavy-seeded bottomland hardwood species, such as oaks and pecans involve seed collection and subsequent planting by hand or machine. Newling (1990) reported that planting works well at most times of year. Because of observed extensive drought-induced mortality of newly germinated seedlings, Sharitz (1992) recommended planting seedlings as a better method of establishing wildlife habitat quickly, even though seeding may cost considerably less. One successful planting technique to obtain mixtures of species involves planting blocks or rows of a single species interspersed with blocks or rows of other species. This approach enhances establishment of slower growing or poorly competing species and allows placement of different species across within-site hydrologic and other gradients.

Riparian wetlands: Plant selection should be done on a site-specific basis, considering the substrate, microclimate, natural water-level regime, plant resistance to erosive stream flows, and dynamics of the riparian community in space and time (Manci, 1989). Plant materials available for revegetation include native seeds available from commercial sources and dormant pole cuttings from adjacent habitats. Most revegetation projects in riparian habitats have used rooted 1-gallon plants grown from cuttings in nurseries, although a few shrubs have been grown from seed. Covering seeds after seeding is essential to most germination and seedling establishment. The success of seeding efforts can be enhanced by use of seed drilling, hydroseeding, or cyclone seeders. Erosion control by means of matting or mulching can provide temporary cover of exposed soils and moderate the effects of rainfall, runoff, and wind. Manci (1989) noted that willow cuttings, which are easy to obtain and less expensive to grow than transplants, can be taken from local sources better adapted to specific site conditions. Carothers et al. (1990) emphasized that all plant material must be protected from desiccation during transport to the restoration site. Because streamflow is a major mechanism for seed dispersal to riparian habitats, controlled flooding may be a feasible method for vegetation establishment. The timing of flooding is critical because the duration of seed viability in some species is short, e.g., one to two weeks for willows (Manci, 1989). Fertilization may be necessary to enhance initial seedling establishment.

Protection

Protection from natural predators and human interference is critical for all types of restored freshwater wetland habitats. Protection may consist of fencing or of vegetated buffer zones. In riparian habitats, fencing is recommended to protect plant material from depredation by beavers, waterfowl, livestock and off-road vehicles. Occasionally, individual trees are fenced separately (Carothers et al. (1990).

Maintenance

It is generally agreed that long-term maintenance of restored or created wetland sites is desirable. Maintenance is site-specific and may involve herbivore control, upkeep of buffer zones, weed control, fertilization, irrigation, and replanting as necessary.

Criteria for Success

All of the literature reviewed, whether concerning oiling or not, focused almost exclusively on restoration and recovery of vegetation. While studies of invertebrate fauna in marine wetlands were rare, such studies in freshwater habitats were non-existent. A few studies mentioned use of restored freshwater wetlands by higher trophic levels, usually referring to avifauna or other wildlife. None of the studies reviewed specifically included evaluation of wetland functional values as a criterion for success, although a number of authors mentioned their importance.

3.2.2.3.1.3 Restoration Actions Following Oiling of Bogs and Fens

Arctic Environments

A number of factors must be considered in planning cleanup or rehabilitation activities in arctic environments.

- There is potential for serious degradation of permafrost (thermokarst) following disturbance. Any surface disturbance will induce thermal degradation of the permafrost and subsequent subsidence, with the result that total environmental injury is increased. Vegetation cannot reestablish itself until the site stabilizes by reaching a new erosional equilibrium (Johnson and Van Cleve, 1976).

- The thick, slowly decaying organic layer, which covers mineral soil in arctic environments functions as a nutrient reservoir and source of insulation, should not be disturbed.

The extremely short growing season of cold-dominated environments results in special problems for the use of introduced plant species in restoration efforts (Van Cleve, 1977; Linkens et al., 1984).

Seasonal effects must be considered in evaluating cleanup activities in arctic environments. For example, crude oil discharged in winter is more viscous and, if the pour point is relatively high, the oil can be scraped from snow and ice surfaces (Wein and Bliss, 1973; Linkens et al., 1984). Absorbent booms may be used to prevent remobilization of oil during snow melt (Linkens et al., 1984) if conditions are appropriate (e.g., small chunks of floating ice could overwhelm and ride over or break the boom). Crude oil with lower pour points will flow according to natural drainage patterns. Summer discharges will move laterally along the permafrost table or water boundary until the lowest level is reached (Wein and Bliss, 1973).

Dyking to contain discharged oil is generally not recommended in arctic environments because of the risk of causing thermokarst in the ice-rich soils. Summer burning of discharged oil and post-discharge cultivation to increase aeration are not recommended for the same reason (Wein and Bliss, 1973).

Considering these factors, passive or no-action cleanup scenarios are preferred whenever possible (Linkens et al., 1984). Linkens et al. (1984) recommend the following actions for restoration and/or revegetation of arctic environments following oiling:

- No action if natural recovery is likely to occur, e.g., if the discharge is small, if the discharge is a spray rather than a point-discharge, and if access to the discharge site is difficult;

- Fertilization is recommended for moderately impacted sites where the root zone is only partly saturated with oil;

- Raking to promote water infiltration and aeration is recommended to increase decomposition of oil in saturated, moderately drained sites;

- Fertilization and reseeding are recommended for heavily impacted or erodible sites;

- Tillage is recommended for accessible, stable sites that are heavily contaminated with oil in the root zone;

- Transplanting is recommended for highly visible sites in which the root zone is heavily contaminated with oil and in which the potential for natural recovery is low; and

- Soil amendment is recommended for highly unstable, heavily oiled sites with low recovery potential.

For oiled sites where revegetation is recommended, a number of factors must be considered, including site conditions, nutrient regime, plant adaptations, plant species, and revegetation methods. These factors are summarized separately below.

Site conditions: The substrate type, climate, thermal regime and topography of the site must be favorable for seedling germination and survival (Johnson and Van Cleve, 1976).

Nutrient regime: Soil nutrients and nutrient requirements of plants must be compatible (Johnson and Van Cleve, 1976).

Plant adaptations: Plants used in revegetation efforts must have both physiological summer cold-hardiness and winter cold-hardiness, i.e., must resist snow abrasion and other stresses. Rhizome regrowth, which provides new stock for revegetation, is an important factor.

Native versus introduced species: Large seed supplies are more likely to be available for introduced species, and these are more likely to require fertilization in arctic and subarctic environments (Johnson and Van Cleve, 1976).

Revegetation methods: Revegetation methods reviewed by Johnson and Van Cleve (1976) include the following items: seedbed preparation, seeding methods, timing of seeding, seed mixes, and fertilization. Seedbed preparation is especially important in tundra environments. Seeding methods include drilling and broadcasting. Johnson and Van Cleve (1976) cite studies showing that seed drilled in rows had germination rates 1.2-7.5 times higher than broadcast seed, depending on species. This result was attributed to improved moisture conditions with drilling. Seed mixes, including mixtures of growth forms, function to increase the variety of seed stocks in order to enhance revegetation over a wide range of conditions and sites. Fertilization was cited as the most important factor for establishment and growth of agronomic species, especially in cases where the organic mat has been removed. Johnson and Van Cleve (1976) reported studies in which fertilization with nitrogen, phosphorus and potassium resulted in marked increases in percent plant cover, biomass, plant height, and vegetative reproduction at arctic sites.

Agronomic grasses and legumes for revegetation should be selected on the basis of reproductive potential, ability to survive several growing seasons, root and top biomass production, rate of plant development, and rate of ground cover development. For a three to four year period in the arctic, only arctic fescue and nugget-Kentucky bluegrass are rated as successful by all researchers (Johnson and Van Cleve, 1976). Little research has been done on introduced woody species, and they are not used widely in revegetation efforts. Introduced grass species in arctic revegetation efforts usually fail after four to five years. Johnson and Van Cleve (1976) noted that such failures are not necessarily bad because the introduced plants may function as nurse species for native plants.

Non-arctic Bogs and Fens

Because of the paucity of published information, recommendations cannot be made for these environments.

3.2.2.3.2 Recovery Times Following Oiling of Freshwater Wetlands

Recovery times of one to two years were reported for cattail marshes in the Saint Lawrence River and Little Panoche Creek, California (Alexander et al., 1981; Pimentel, 1985). A California scrub-shrub wetland was said to have recovered within six months (Kemerer et al., 1985). However, in all of these cases, the vegetation was not completely killed by the discharge. Recovery times would have been much longer if all the vegetation had been destroyed. Recovery times of oiled wetland forests were not estimated. However, if mature forest vegetation were killed by oiling, recovery times would be on the order of several to many decades.

Recovery times are long in arctic and subarctic taiga and tundra environments, occurring over a timescale of years to many decades (Hunt et al., 1973). Recovery in these habitats is greatly affected by the rapidity with which oil penetrates the soil. An oil-free top layer of soil appears to be required before vegetative recolonization and recovery can proceed (Hutchinson and Freedman, 1975). Wein and Bliss (1973) reported that in areas characterized by dwarf birch, willow and heath shrubs, considerable regrowth from latent buds occurred after three to five years, provided the discharged oil was not highly toxic. Times for revegetation to occur are much longer in arctic than in subarctic environments because of lower summer temperatures and a shorted growing season; a difference on the order of 1,000 years is possible. In the arctic, revegetation does not seem able to prevent thermokarst and may only help restore thermal balance after many years (Johnson and Van Cleve, 1976).

3.2.2.3.3 Monitoring of Freshwater Wetlands

Ideally, oil-impacted wetland habitats should be monitored over a time period appropriate to document recovery. The timescale of monitoring will be discharge- and site-specific. Components of monitoring programs for freshwater wetlands are the same as those described for saltmarsh and mangrove habitats in sections 3.2.1.1.3.3 and 3.2.1.2.3.4.

Ideally, the minimum monitoring time is equivalent to the time to maturity of the dominant vegetation. This will generally be on the order of a few years in riverine marshes, but may be decades in subarctic bogs and fens and temperate forested wetlands. If pre-incident baseline data are not available, unoiled reference sites must be established. Monitoring surveys should be designed so that temporal changes can be resolved statistically. Measurements will by definition focus on vegetation, and should also include invertebrate fauna. The oil content of substrate should be measured in sediment cores.

3.2.2.3.4 Recommendations for Future Research

Freshwater habitats are not as well studied as saltmarsh and mangrove habitats in terms of recovery from oiling. Basic information concerning how soon to plant after oiling is not available. Future research needs include development of non-destructive response methods, including bioremediation, to oiling of all types of freshwater wetlands. The time scales of recovery of functional values, including nutrient pools, biomass production, and trophic transfers, need to be better understood for assessing the need and actions chosen for restoration. For example, sediment removal and replacement may severely disrupt these functions such that recovery is prolonged over that which would occur naturally even in the presence of contamination.

3.2.3 Vegetated Beds

3.2.3.1 Macroalgal Beds (Estuarine and Marine)

3.2.3.1.1 Intertidal Macroalgal Beds

Intertidal macroalgal beds are an essential component of the rocky intertidal and inseparable from that habitat. Refer to section 3.2.6.1 for a discussion of intertidal rocky shores.

A short discussion is given here of restoration work proposed for *Fucus* beds in response to injuries resulting from the *Exxon Valdez* oil discharge and the ensuing response. Stekoll (1993) has noted that there was significant removal of *Fucus gardneri* from the mid- and upper intertidal zones in areas oiled by the discharge (due more to response treatment, than to oiling). Due to the limited dispersal ability of the *Fucus* and the harsh environment of this habitat, a very slow recovery is anticipated. It appears to be recovering faster at the exposed than at the sheltered stations, but is still far from recovery. Not only is *Fucus* reduced in number and size, but the few remaining plants of reproductive size have fewer fertile receptacles and are thus less fecund (EVOS Trustees, 1992e). The *Exxon Valdez* Oil Spill Trustees (1990) proposed a restoration feasibility study for restoring *Fucus* to the intertidal and hopefully thereby speeding the restoration of its associated community. The elements of this project are to document natural *Fucus* recruitment in areas exposed to oil, assess feasibility of actively restoring *Fucus* to these areas, develop techniques for the large-scale growth of *Fucus* seedlings, compare the effectiveness of seeding *Fucus* versus transplanting it, and evaluate the costs for a full-scale *Fucus* restoration project (EVOS Trustees, 1990c). This project may or may not be pursued depending on selection from the full list of proposed studies and uses for the *Exxon Valdez* settlement funds.

3.2.3.1.2 Kelp Beds

Much of the available information on kelp beds is concerned with the giant kelp forests off the coast of California dominated by the kelp *Macrocystis pyrifera*. The emphasis here will be on that habitat, although passing reference will be made, as appropriate, to other species. As with other organism-defined communities, a kelp bed is more than seaweed. It is a complex community made up of many species with many interactive functions, which rely on the structure, productivity and physical properties of the kelp and its presence in that environment. These aspects of the kelp forest are summarized by Foster and Schiel (1985).

3.2.3.1.2.1 Oil Discharge Effects on Kelp Beds

There are no known cases of kelp bed restoration in response to injury from oil discharges. A review of injury and natural recovery from historic oil discharges will be quite brief. North et al. (1964, cited in Foster and Schiel, 1985) studied the impact of the *Tampico Maru* oil discharge on a *Macrocystis* bed in Baja California. Dramatic mortalities of invertebrates resulted, with less obvious injury to the kelp. Five months after the discharge there was good kelp growth which eventually increased in area over prespill coverage, apparently in response to the lowered grazing pressure by the reduced animal community. Major macrophyte grazers (sea urchins and abalone) were absent for more than two years after the discharge and species richness continued to increase for ten years, suggesting a continuing recovery process (Johnson and Pastorak, 1985).

The Santa Barbara oil discharge of 1969 resulted in oiling of the water surfaces over kelp beds and in many deaths of birds associated with the kelp. There was also an observed decline in mysids. Otherwise, little injury was observed to the kelp, fish, or invertebrate communities (Foster and Schiel, 1985).

Other references to impacts of discharged oil on kelp are anecdotal or uninformative. Thus, there is little oil discharge related information on which to base conclusions. Johnson and Pastorak (1985) offer some useful observations. It appears that the kelp itself may recover rapidly (one to a few years) but that the other elements of the community may take longer to recover. Annual forms of kelp (e.g., *Nereocystis*) can be expected to recover more rapidly than perennials (e.g., *Macrocystis*). Most importantly, they observe that a kelp bed is really one form of alternate stable states for a rocky bottom subtidal area. The natural return to a previous state depends to a large degree on the impact on other members of the community such as the grazers and their predators (Johnson and Pastorak, 1985).

3.2.3.1.2.2 Restoration of Kelp Beds

Historically, losses of kelp beds have been attributed to a number of causes. Ever-increasing sewage discharges to the marine environment off the California coast has increased sedimentation, and turbidity, and added potentially-harmful toxics. A combination of changes associated with *El nino* events lead to warming of the water, decreased nutrients and an increase in severe water motion that together may lead to loss of kelp beds. Changes in faunal populations, whether due to over-fishing or to natural population cycles, can lead to overgrazing which may reduce kelp beds (Schiel and Foster, 1992). Schiel and Foster (1992) point out that the kelp beds along the coast of California have undergone considerable, apparently natural, variations in coverage over the past century and that this natural variation should be allowed for in interpreting success or failure of transplant efforts. They express some skepticism over the "success" of some past transplant efforts, noting that there has been inadequate consideration of natural variation and its causes in accounting for success and failure.

Kelp bed restoration may consist of transplantation or seeding, predator or competitor control, or some combination of these tactics (Wilson, Haaker and Hannan, 1977). Transplantation is the primary stratagem, which has been attempted with varying success in a number of places. The intention of a transplant program is not to replace a kelp bed, but rather to provide sufficient seed material in the environment to allow it to naturally reproduce and spread. Several approaches have been tried. Whole plants--adults or juveniles--may be pried from their substrate and transported (with appropriate precautions) to their transplant site and attached in place. Several techniques are employed for attachment. Where appropriate, they may be held in place by attaching the hold fast to a solid substrate with a rubber ring (Wilson, Haaker and Hannan, 1977). Where sea urchin grazing is a potential problem, this approach is altered by attaching the hold fast to a float and suspending the float a short distance off the bottom with a nylon line. (For more discussion of techniques, see Chapter 2.)

There has also been some experimental work with dispersal of spores or laboratory-raised embryonic sporophytes. (The sporophyte is the life history stage which becomes the large, obvious kelp plant. The short-lived, alternate, gametophyte stage is not generally seen.) This approach, though promising, remains experimental. While it allows very large numbers of potential plants to be released to the environment, they are very sensitive to environmental conditions for successful settlement and growth and the tiny plants are at a stage very vulnerable to grazers and competitors. Effective restoration using this method requires numerous seeding events over a period of time to ensure some of the plants an appropriate window of environmental conditions for settlement and growth (Schiel and Foster, 1992). It may also require an aggressive program to control grazers and competing plants.

Where suitable substrate does not exist, it may be provided. This has taken several forms. The Los Angeles Harbor Department, as part of a mitigation plan, carried out a kelp transplant project in Los Angeles Harbor in order to enhance the wildlife resources there (Rice, 1985). In order to provide attachment points for the transplants, 12 meter lengths of chain were weighted in place perpendicular to the breakwater. Transplant stock was attached to floats tied to the chain with nylon line.

Artificial reefs have been constructed in at least two places to mitigate possible losses from power plant activity. The results, which highlight the need for proper consideration of the conditions that lead to development of a healthy kelp bed, are discussed by Schiel and Foster (1992). The Pendelton Artificial Reef, near San Diego, was unsuccessful for eight years. Its development probably suffered from a number of features which may be summarized as poor site selection and reef design. Its eventual success was probably due in part to a combination of more favorable environmental conditions. Another artificial reef, constructed four years after the Pendulum Reef, did not have these problems, having a design more appropriate to kelp bed development and being located closer to other kelp beds. Kelp was growing on this reef within six months (Schiel and Foster, 1992).

Kelco, Inc. (1990) has developed techniques for directly restoring kelp on sand bottoms. While kelp beds generally develop on hard substrates and may in fact be limited in extent by surrounding sand bottom (Schiel and Foster, 1992), they are in some places found growing on sand bottoms. Plants grow attached to large rings of old hold fast material called growth centers. It appears that conditions conducive to growth in sand and development of these growth centers are not common. When the kelp beds growing on the sandy bottom off Santa Barbara county went into decline beginning in 1982, the kelp was not able to re-establish itself. Kelco (1990) proposed that the primary problem limiting the regrowth was a lack of growth centers. They have developed a series of restoration techniques that have been tested on a pilot scale. They constructed "mushroom anchors," consisting of a concrete anchor with a flat surface and a convex base with rebar handles (which then serve as points of attachment for growing kelp hold fasts) and a transplant attachment structure. These artificial growth centers (AGCs) are deployed on sand bottoms at a density equal to natural growth center density. The AGCs settle into the sand such that the upper 2 to 5 cm of the surface remains exposed. *Macrocystis* plants recruited to these structures within a year when AGCs were deployed near existing kelp beds. Kelco (1990) also used these AGCs as transplant anchors. Juvenile plants were attached to the attachment structure and spread out over the sand bottom. This did not prove to be successful due to grazing problems, and perhaps poor water quality. They did, however, have a later natural set of plants on the AGCs.

A third approach Kelco (1990) has used on sand bottoms is to "staple" plants in place. Barbed rebar staples are used to reattach loosely-attached plants in place to help maintain their hold on the substrate and form new growth centers.

An important aspect to kelp transplants relates to planting density. Small sparse replanting efforts have a poor record of success, at least partly due to grazing problems. If frond density is too sparse, the grazers (fish and sea urchins, mostly) may consume the plants to a point where they cannot survive. Several transplant projects have suffered this fate (Schiel and Foster, 1992; Kelco, 1990; Rice, 1985; North and Neushal, 1968). Transplant programs must be sufficiently large to dissipate the effects of these grazers over many plants or grazers must be controlled.

Most grazer control concerns the effects of sea urchins on kelp. A variety of techniques have been employed to control sea urchins. These include using divers to smash them with hammers, collecting and destroying them in other ways, or applying quickline (CaO) which kills them in place (Schiel and Foster, 1992). Since sea urchins (at least some species) are now considered a valued resource, these techniques are now inappropriate, and, in fact, will not be necessary in many places where they are fished. Schiel and Foster (1992) question the certainty of the relationship of sea urchin control to kelp bed success, pointing out that in some cases these successes were accompanied by amelioration of other environmental factors. They point out (citing Ebeling and Laur, 1985) that there may be a natural transition from communities dominated by kelp to those dominated by sea urchins, and back, in five years. Nevertheless, it seems probable that in any restoration attempt, during the period when the new kelp is sparse, some control of sea urchins will be needed to allow the plants to get started and to reproduce. Control of grazing fish may prove more problematic. Gill nets and hardware cloth protection structures have been employed but do not seem to provide satisfactory solutions (Schiel and Foster, 1992; North and Neushal, 1968).

3.2.3.1.2.3 Kelp Bed Restoration and Recovery: Summary and Conclusions

The small history of oil discharge impacts on kelp beds implies that consideration of direct restoration of kelp is unlikely. It is more probable that where injury occurs, it will be to the large and diverse animal community that lives in this habitat. There is a poor record for restoration of any of these animal species. Thus, for the foreseeable future, natural recovery will be the most viable restoration alternative. Monitoring of this recovery should include as an important component assessment of the condition of the kelp bed. If injury to the kelp bed fauna were to selectively harm the predators, kelp grazers might then expand their populations and overwhelm the kelp.

Where there is extreme injury to the kelp, to the point where active restoration is contemplated, there must be careful consideration of specific conditions at hand. Natural recovery may still be more appropriate. Schiel and Foster (1992) state it most clearly: "Most evidence to date suggests that natural recovery swamps efforts at restoring." Efforts might then be best directed at assisting natural recovery through control of grazers and competitors in the early stages of the re-establishment of the kelp bed. In monitoring recovery, there must be careful attention to the conditions that are conducive to good growth and the recognition that there are natural cycles of kelp beds that are still only partially understood. Wilson, Mearns and Grant (1980) and Schiel and Foster (1992) point out the considerable importance that improving natural conditions have had on the apparent success of restoration efforts.

Future research is still needed on the conditions that are conducive to kelp bed maintenance and growth. The causes of past successes and failures and the conditions required for a successful restoration are not always clear. Research should also be undertaken on optimal conditions for survival of spores, and settlement and survival of sporophytes, to the end that these may provide viable means of reseeding kelp beds.

3.2.3.2 Seagrass Beds

Seagrass beds, whether tropical or temperate, provide important, highly productive habitats in marine coastal environments.

Zieman and Zieman (1989; citing Wood et al., 1969, and Zieman, 1982) list the general environmental functions of seagrass beds as follows:

- High production and growth. Rapid growth allows them to exert a potentially large influence on local environments;

- Food and feeding pathways. Seagrass is an important source of food both directly and as detrital material after it dies. Some of this production may be exported considerable distance;

- Shelter. Seagrass beds provide important habitat for some or all life stages of a variety of animals;

- Habitat stabilization. By slowing currents through the bed, seagrass leaves promote sedimentation. This current-retarding action, as well as binding by roots and rhizomes, stabilizes the sediment against erosion; and

- Nutrient effects. Seagrass and the seagrass ecosystem are active at all levels in the nutrient cycles of their surrounding environments.

These functions are not incidental to the subject at hand. Most importantly, they point out that a seagrass bed is not merely a field of a single species, but rather a complex system made up of many components including benthic algae, epiphytic plants and animals, epibenthos, infaunal benthos and nekton (Phillips, 1984). In addition, the seagrass bed interacts with the surrounding environment to provide additional services to species as disparate as reptiles and birds. Clearly, to evaluate impacts to a seagrass bed from an oil discharge, or from any other source of injury, it will be necessary to look at more than the grass itself. An accurate measure of impacts and recovery from injury will only be possible through consideration of a variety of the elements making up this ecosystem. This is a very demanding task that has never been carried out for a seagrass restoration. We will instead have to rely on indicators of recovery, such as the seagrass itself, and make suppositions about the extent to which this reflects the whole system.

3.2.3.2.1 Oil Discharge Effects on Seagrass Beds

Although there are records of oiling of seagrass meadows, there is no known instance of restoration of seagrass beds, temperature or tropical, in response to injuries from an oil discharge. As a result, this review will concentrate on impacts and on rate and measures of natural recovery. Temperate and tropical grassbeds are discussed separately.

3.2.3.2.1.1 Temperate and Subarctic Seagrass Beds

Temperate and subarctic seagrass beds are represented largely by the eelgrass *Zostera marina* in the United States, through other species are occasionally found. Studies of eelgrass ecosystems and characterization of Pacific northwest and Atlantic coast eelgrass meadows are summarized by Phillips (1984) and Thayer and Fonseca (1984), respectively.

While no instances of seagrass bed restoration in response to injuries from an oil discharge are found in the literature, there are several accounts of conditions in eelgrass beds following oiling. In some cases, these studies include follow-up observations to evaluate natural recovery. However, none of these monitoring studies are very rigorous. The studies suffer from two problems inherent to the system. First is the complexity of the ecosystem that makes it an almost insurmountable task to consider all the possible elements of system recovery. The other is the fact that oil discharges rarely occur in locations where extensive pre-incident data for affected environments already exist. Control or reference sites have to be selected that may or may not fairly represent the original condition of the injured site.

An early observation of oil impacts on temperate seagrass beds was for the *M.C. Meigs* grounding on the Washington coast in 1972 which oiled an intertidal bed of *Phyllospadix scouleri* or "false eelgrass" (Clark et al., 1975). This reference notes that heavy oiling of this bed resulted in high retention of oil, but makes no mention of injuries either to the *Phyllospadix* or to its associate community. Consequently there is no information on recovery from this discharge.

Foster et al. (1971) noted injury to the intertidal surfgrass *Phyllospadix torreyi* resulting from the Santa Barbara oil discharge in 1969. Grass blades readily took up oil and held it. Where this occurred, the blades eventually turned brown and disintegrated. Oil did not stick to most of the nearby algae nor to the low intertidal and subtidal plants that appeared uninjured. The rhizomes of the surfgrass remained covered with sand and it was suggested that the grass might recover from the impacts through vegetative growth. There was no other information on recovery.

The *Amoco Cadiz* discharge off the Brittany coast provided an opportunity to study the impacts of an oil discharge on eelgrass beds in the path of the discharge. *Zostera marina* beds at Roscoff, France were monitored. Estimates were made of the production and biomass of eelgrass and the faunal composition of the grassbed community. The initial results of the production and biomass studies are summarized by Jacobs (1979). Unfortunately, there is no published follow-up to this aspect of the study. The monitoring of community composition had started only six months prior to the discharge, thus limiting the precision of any conclusions that may be made. It was, nevertheless, a unique opportunity in that some truly representative pre-incident data existed for the area of discharge impact. The effects of the discharge on the eelgrass community are discussed by Jacobs (1980) for the benthic infauna and den Hartog and Jacobs (1980) for the mobile benthos.

The subject eelgrass beds were hit by oil on March 20, 1978. The oil remained for weeks, covering the beds at low tide and loosening and floating off at high tide. Despite this heavy oil coverage, the impacts to the grass itself were not severe. In April and May, 1978, especially in the shallower study area, there was a blackening of the leaves and presence of transparent areas on them. These leaves were shed, but the plants were still alive. Production was judged to have continued normally and the general structure of the eelgrass beds was not altered (Jacobs, 1979, 1980; den Hartog and Jacobs, 1980).

A decrease in numbers of individuals and species was immediately apparent in the benthic infauna. Results in the shallower study area proved difficult to analyze due to natural changes in the bed. In the deeper bed these faunal changes were most apparent as a disappearance of amphipods, tanaids, and echinoderms and a reduction in numbers of gastropods, polychaetes, and bivalves. By the end of 1978, numbers of individuals had returned to levels present a year earlier, but diversity continued to change. The echinoderms were slow to recover and none of the filter-feeding amphipods had returned. However, compared with some other habitats, it was concluded that the eelgrass community suffered relatively mild impacts since eelgrass blades and rhizome mat may have provided a protected habitat, reducing the impacts of the discharge on its residents (Jacobs, 1980).

Total numbers of individuals and species of mobile benthic fauna also decreased immediately following the discharge, an effect more evident a month later. Numbers of individuals increased throughout the following year but did not reach levels equalling those of a year earlier, and species numbers remained lower than before the discharge. Gastropods were not adversely affected. Cumaceans, tanaids and echinoderms had nearly recovered within a year. Amphipods were severely affected. There were 26 species of amphipods in the bed preceding the discharge, of which 21 had not returned a year later (den Hartog and Jacobs, 1980).

Houghton et al. (1991a,b; 1993a,b) evaluated the impact of the *Exxon Valdez* oil discharge and consequent response efforts on the shoreline and eelgrass beds offshore of treated, untreated and unimpacted shorelines. This study only considered eelgrass-specific impacts in the seagrass beds and did not evaluate impacts on other elements of this community. There appeared to be no impact by exposure to oil on the vegetative structures or processes, but there were some measurable impacts on reproductive processes. A year after the discharge, this effect (i.e., low flowering shoot density) was generally evident at all oil-impacted sites. Two years later, only those sites offshore of oiled shoreline that were subjected to high-pressure hot water washing showed this effect. This presumably reflected incorporation of hydrocarbons into the sediments through the washing process.

Duval et al. (1989) described some results of the *Nestucca* discharge off Vancouver Island. It was noted by divers that oil in the water column moved through the kelp freely but adhered to the eelgrass. Some eelgrass beds were sufficiently oiled that the grass was removed to prevent geese from eating it. The oil might also have contaminated the marine food web.

3.2.3.2.1.2 Tropical and Subtropical Seagrass Beds

Seagrasses in the southern United States are represented primarily by three species: *Syringodium filiforme* or manatee grass, *Halodule wrightii* or shoal grass and *Thalassia testudinum* or turtle grass, as well as by two species of *Halophila* and by *Ruppia maritima* (Zieman and Zieman, 1989). The biology, ecology, productivity, and dynamics of seagrasses of the west coast of Florida are summarized in some detail by Zieman and Zieman (1989). As with eelgrass beds, no published accounts were found of tropical seagrass bed restoration in response to injuries from an oil discharge, although there is a significant literature for seagrass restoration from a variety of other impacts. Several accounts are given in the literature of impacts of oil discharges on tropical seagrass beds and some of these include information on natural recovery. Again, however, none of the studies reviewed were adequate to fully evaluate restoration of the communities involved to their original state.

Nadeau and Bergquist (1977) describe the effects of the 1973 *Zoe Colocotronis* oil discharge in Puerto Rico on a variety of communities. These communities included sublittoral *Thalassia* beds and flats. Quantitative surveys were made in several affected *Thalassia* beds as well as in unoiled control sites one week and thirteen weeks after the discharge. Epifaunal and infaunal benthos were evaluated. There were also follow up visual surveys. There was a considerable initial die-off of animals seen in some of the affected areas and this was quantified in the surveys for one of the three beds studied. Thirteen weeks later, diversity was increasing but still low, except in one area. It was only in these latter flats that grass injury was noted. Blades were killed and the rhizome matrix was exposed by erosion due to the loss of protecting grass blades. A year later, growth was underway. Three years later, there was renewed plant growth with sediment deposition. Repopulation of lost fauna at the other beds was noted one and three years later, except for the queen conch, a commercial species that may have been reduced by fishing pressure. The ability to conclude much from these studies is limited by the inherent variability observed both among oiled areas and between oiled and control areas. No statistical tests were shown and it is unlikely any could have been successfully applied.

Chan (1977) described some of the effects of an oil tanker discharge in the Florida Keys in 1975. The area has extensive seagrass cover, but there was no oiling observed of attached species (*Thalassia*, *Diplanthera* = *Halodule*, and *Syringodium*) following the discharge. However, dead grass (apparently unrelated to the discharge) picked up oil and was washed onto the shore. The only recorded evidence of injury was a large die off of pearl oysters (*Pinctada radiata*) which was likely due to the oil contamination.

On April 27, 1986, there was a major oil discharge in Bahia las Minas on the Caribbean coast of Panama. Impacts on a variety of communities, including extensive intertidal and subtidal *Thalassia testudinum* meadows are described by Jackson et al. (1989) and Keller and Jackson (1991). This event provided an important, unique opportunity for oil discharge impact assessment in tropical environments since some of the affected areas had been the subject of ecological studies for 18 years preceding the discharge. Unfortunately, these studies did not include the subtidal seagrass communities, for which there was little data. Thus, evaluation of impacts for these communities was based on comparison of oiled and unoiled communities from the same region, which limits the confidence in any conclusions that can be made. The injury was heaviest in the intertidal region where entire beds of *Thalassia* were killed in some of the worst-hit areas. Oil soaked into the sediment, killing the rhizomes, which eventually rotted away. The unprotected sediment eroded to bare rock and has not recolonized since (Cubit and Connor, 1993). However, subtidal seagrass survived everywhere. In the heaviest hit intertidal grass beds, there was browning of the leaves and heavy fouling by algae for several months following the discharge. Some of the animals living in these subtidal beds were significantly affected. Amphipods, tanaids, brachyurans (crabs), and polychaetes were significantly less abundant in oiled beds than in control beds four months after the discharge. Abundances of ophiuroids, bivalves, burrowing shrimp, and gastropods were not significantly different, although their numbers were lower.

Abundance of most taxa increased in all areas, oiled and unoiled, over the following four months. Amphipods, tanaids, and ophiuroids showed poor recovery in oiled areas (Jackson et al., 1989).

In oiled subtidal beds, seagrass biomass was reduced compared to control sites just after the discharge, but was equal to control sites seven months after the discharge. In the intertidal, however, the shoreward edges of the beds were receding three years after the discharge (Keller et al., 1991). Longer-term faunal impacts were not clear. A year after the discharge, most infauna were similar in control and oiled subtidal sites. Epifauna and nekton were more variable, some shrimp were more common, while others were less common at oiled sites. Small fish were generally less abundant (Keller et al., 1991).

Only one experimental study of oil impacts on seagrass beds was found. Ballou et al., (1987) carried out a two and one half year field experiment on the impacts of a severe fresh-oil discharge, with and without dispersant, on mangroves, seagrass and corals. The seagrass was a subtidal *Thalassia testudinum* bed. Sites were sampled twice for prespill data, eight months and one week, prior to the oiling. Sites were then oiled for two days, with and without dispersant application, and monitored for 20 months. Neither treatment showed any significant effect on the growth rate or blade areas of the seagrass. Sea urchins were heavily affected at the dispersed oil site but reappeared a year later. They only slightly decreased at undispersed oil sites. The results with infauna sampling were so variable, both for density and for diversity that no pattern could be discerned between sites or over time (Ballou et al., 1987).

Clearly, the above brief history is inadequate to draw definitive conclusions regarding impacts of oil on seagrass beds, temperate or tropical. However, oil discharges do not appear to be especially injurious to seagrass, while the community therein may be quite sensitive. While recolonization by resident fauna was not well studied, there is a prevailing suspicion that if the structure (the seagrass itself) is provided, it will recolonize rapidly from surrounding environments (e.g., Fonseca et al., 1990). It is, however, extremely important not to disrupt the system physically. The root-rhizome mat formed by seagrasses is an essential structural element of the seagrass bed, and injury to this component could considerably slow recovery. The fact that intertidal *Thalassia* beds may be killed outright by a heavy oil discharge, as observed by Jackson et al. (1989) and Keller et al. (1991), indicates that evaluation of direct restoration actions is needed. There is little information to evaluate the natural recovery of seagrass beds and no work was discovered on restoration of these habitats after an oil discharge. The following table, taken from Zieman et al. (1984), summarizes these conclusions for seagrass beds in general and oil discharge impacts:

Damage Level	Plant Effects	Associated Community Effects	System Fate	Recovery Time	Restoration Indicated
1	No visible damage	Possible faunal damage	Natural recovery	Weeks to years	No
2	Leaf damage and removal	Faunal damage may be extensive	Natural recovery likely	6 months to years	No
3	Severe damages to rhizomes	Faunal damage is likely extensive	Natural recovery slow or unlikely	5 years to decades	Yes
4	Severe system damage	System completely altered	Return to same state not possible	?	No

They conclude that management efforts should be primarily focused on limiting the injury and maximizing the probability of natural recovery.

3.2.3.2.2 Restoration of Seagrass Beds

Seagrass bed restoration has been undertaken in many places for a variety of reasons not related to oil discharge injury. Changes in the environment may increase currents or waves that can cause grassbed changes. Boat traffic may also contribute to this problem. Increased turbidity may reduce light to the bed, as may eutrophication effects. Grassbeds uniformly need high light levels to thrive and will die out where water clarity becomes significantly degraded. Thermal pollution has led to grassbed destruction, as have bioturbation, storm scour, and overly aggressive fishing efforts. In some areas the most destructive causes of grassbed loss have been dredge and fill operations.

While it is believed by some that seagrass bed restoration is effective and should be considered a useful option, others consider it to be of highly questionable reliability. Dial and Deis (1986) point out that seagrass bed restoration is still experimental. There are questions about the best methods to use, and reasons for success or failure are not always clear. Thus, restoration or replacement by seagrass creation should be considered experimental (Fonseca, 1989).

3.2.3.2.2.1 Location for Seagrass Restoration

The most important issue in establishing a program of seagrass restoration is appropriate location. This issue is widely considered of overwhelming importance, and even where a restoration is proposed for a site where seagrass previously grew, the principles inherent in this issue should be borne in mind by those planning the restoration. If an area does not presently support seagrass growth, there is probably a reason for that fact and there should be sound justification for attempting to plant there. This principle that seagrass should only be planted where it is known (historically) to be able to grow has been restated by several authors (Fonseca et al., 1987a; Curtis, 1991; Fonseca, 1992; Kirkman, 1992).

This principle also applies on a smaller scale as well. Grassbeds will often have open areas in them. This patchiness often has a real cause and attempts to plant in these areas to mitigate losses elsewhere may lead to failure. It may be that underlying substrate at the open areas is inappropriate or that there are hydraulic reasons for the open spaces. In areas of high currents it is natural for open spaces to develop in grass beds. Fonseca (1989) and Fonseca et al. (1987b) also observe another important point, that these open spaces are habitat as well. They contribute to the overall diversity of the environment and probably to the productivity of higher trophic levels.

Given this skepticism over replacement planting, there is still a belief that it is possible. Fonseca et al. (1987b) indicate that with appropriate planning, a site such as a dredge fill area may yet be made suitable for seagrass growth. Fonseca (1992) provides a list of preferred restoration sites that attempts to optimize the probability of mitigating injuries while minimizing the loss of alternate habitats. In order, one should preferentially restore in an area where seagrass once grew where conditions suitable for growth have returned, a dredged or filled area where seagrass once grew that can be returned to original elevations, other areas of dredge and fill, or converted upland areas zoned for development (Fonseca, 1992).

3.2.3.2.2.2 Environment for Seagrass Restoration

In planning a seagrass restoration, there must be a meticulous consideration of the environment of the habitat into which the restoration is to occur and its suitability for seagrass growth. These include physical, chemical and biological factors. In a comparison of transplant success between two geographically separated areas, Fonseca et al. (1987a) considered the following factors: temperature, salinity, light attenuation, depth, hydraulic regime, sediment type, sediment fluctuation, sediment depth, and biotic disturbance. Most or all of these factors have proven (or been suspected) to be important in the success or failure of seagrass bed growth (Fonseca et al., 1987a). The specific factors most conducive to growth of a given seagrass species are not fully understood, such that the best one can do is simulate the environment in which the grass is known to grow, with extra attention to those variables believed to be important (Thayer et al., 1985; Fonseca et al., 1987b). The ideal, of course, is to replant where the loss has occurred. The principles still apply, however. Conditions that led to the loss must in some way have terminated or ameliorated before restoration can be initiated. If the grassbed loss has caused an appreciable change in the environment, the opportunity may be lost. Seagrasses bond sediment and reduce turbidity. If the loss of a grass bed were to result in excessive erosion and accompanying high turbidity, it may no longer be possible to grow grass at that site (Thorhaug, 1986). It may be possible, however, to adjust water depth to an appropriate level with fill (of a suitable texture and chemistry) at sites where grass once grew (Fonseca et al., 1987b) or to add sediment to edges of existing depth-limited seagrass beds (Curtis, 1991) to provide appropriate substrate and depth in areas believed to have the best chance of providing a suitable environment for growth.

The restoration effort itself may involve alteration of the environment. *Halodule wrightii* is a pioneering species in tropical and subtropical areas that establishes itself easily and grows rapidly. *Thalassia testudinum*, on the other hand, is a climax species which takes much longer to establish itself and flourish. Some restorations have sought to use a "compressed succession" (Derrenbacker and Lewis, 1982; Holtze, 1986) involving initial planting of *Halodule* to stabilize the environment with a simultaneous or follow-up planting of *Thalassia* to encourage the ultimate dominance of the preferred climax species (Fonseca, 1992).

Another important environmental variable is season. While seagrass restoration may take place year round in some areas, there is a seasonal component to its growth, and transplanting will be most practical and most successful, if this is kept in mind. Seasonality will affect the availability of transplant material (Fonseca, 1992). There are appropriate planting times and tolerance ranges of plants to environmental variables (Fonseca, 1989). Important seasonal components not related to the grass itself which should be considered include spawning cycles of local fish and nesting by nearby birds (Fonseca, 1992).

If source material for transplanting is acquired from some geographically distant site, it may require some acclimation period to the new conditions to ensure survival (Boone and Hoeppel, 1976). There appear to be physiological races of at least of some seagrass species. Plants from different environments display differing growth and response to environmental conditions (Durako and Moffler, 1981).

3.2.3.2.2.3 Methods for Seagrass Restoration

There are a variety of planting methods that have been tested with varying success in seagrass restoration. While each restoration may have some special variation, the list of methods can be reduced to a simple one:

- Plugs;
- Turfs;
- Individual mature plants; and
- Seeds or seedlings.

Numerous generalizations may be found in the literature about the usefulness, reliability, or applicability of various approaches. Some of these are contradictory. Each restoration will ultimately be designed to address a specific situation and that restoration should incorporate the best available method with those specifics in mind.

Plugs

Plugs are sections of grassbed including blades, roots, rhizomes and the sediment itself that are extracted whole from the donor bed and transferred to the transplant site. Typically these are collected with a 10-20 cm coring device pushed about 20 cm into the sediment. A posthole digger may also work. A corresponding hole has to be made in the transplant bed to accommodate the transplant. Plugs are not generally anchored, but biodegradable pots have been used by a number of workers to transfer the plug, provide a discrete product to place into the transplant bed, and act as an anchor of sorts. Thorhaug (1986) also reports that cement plug collars or chicken wire have been used to anchor plugs. Plugs have a generally good record of success since they disturb the transplanted material minimally and leave it firmly set in the sediment it grows in. It is not, however, widely favored where actions exist. It can be very expensive, involving considerable labor to transport huge masses of sediment and it can leave the donor bed injured as a result of the extractions. It has been identified by some, however, as one of the only methods that has been successful for many species of seagrass (Holtze, 1986)

Goforth and Peeling (1979) transplanted a 1.62 hectare site with eelgrass (*Z. marina*) using 20 cm plugs in perforated biodegradable fiber pots. Surviving plugs at the intertidal site increased rapidly in area, revegetating the site, and regrowth in the donor bed was reported to have obscured evidence of plug removal in a single growing season. Subtidal transplants, however, survived poorly, due probably to heavy growth of *Gracilaria* (an alga) and the resulting shading of transplants. Therefore, the authors pointed out the importance of measuring irradiance at the proposed transplant site and of carrying out pilot studies where conditions are questionable (Goforth and Peeling, 1979). Pilot studies they had carried out had demonstrated that transplant survival varies with plug size but did not anticipate the shading problem.

Phillips (1982) summarized eelgrass transplant techniques, observing that at its range extremes, *Zostera* works best transferred in its own sediment, but other methods work well in between the range extremes. Thayer and Fonseca (1984) conclude that plug transfer of *Zostera* has all the disadvantages already discussed without appreciably aiding survival and has not been reliable in high-current areas. Curtis (1991) reported success transplanting *Zostera* in plugs to low-current areas and commented on its difficulties. He reported problems transplanting in biodegradable pots.

Lewis and Phillips (1981) summarized some seagrass transplant projects in the Florida Keys. They report that plugs give the overall best results and that *Thalassia* survived best of the three major seagrasses when transplanted with plugs. However, Thayer et al. (1985) conclude that there has only been limited success with transferring *Thalassia* with its sediment.

In a pilot study in Biscayne Bay, Thorhaug (1985) reported poor results with plug transplants of *Halodule*. She reported some success with a modified plug technique with *Thalassia* in a follow-up study (Thorhaug, 1987). Large 2 m x 1 m "sods" were extracted from a sacrificial seagrass bed scheduled for beach fill cover. These were covered mostly with *Thalassia* with some *Syringodium*. The large pieces were then subdivided and planted by divers. Seventy percent survival was reported for areas not affected by hurricanes that year (Thorhaug, 1987).

Fonseca et al. (1987a) consider plug transfer of *Thalassia* to be a method of last resort since there are potentially such long-term affects to the donor bed. When it is necessary, only low-energy *Thalassia* donor beds should be used to prevent migrating scour areas and the holes created should be replanted with *Halodule* to stabilize the sediments.

Turfs

Turfs are a less well-defined medium than plugs. There probably is some overlap in what various authors refer to as turfs or plugs. A turf is a piece of intact seagrass, blade, rhizome and roots, with sediment. It is what one might dig out of the grass bed with a shovel. Thus it is a sort of shallow plug, more suitable for species with shallow root/rhizome systems (i.e., *Halodule* but not *Thalassia*). A variety of anchoring methods have been used with turfs to hold the new material in place until it gets established. As it is quite similar to plugging, transplantation by turfs has many of the same advantages and disadvantages.

Individual Mature Plants

Shoots and sprigs are alternate ways of referring to individual mature plants or some part thereof that are cleaned of sediment and planted individually or in clusters of planting units (PUs) that will usually include some type of anchoring device. Phillips (1974) had good success with eelgrass turions. He removed the plants from the sediment with as much rhizome material as possible and attached these shoots to pieces of pipe with rubber bands. These were then buried in 10 cm deep trenches.

Homziak et al. (1982) washed shoots free of sediments and wove them into paper and plastic meshes that were then anchored with steel pins. This appeared to be a successful transplant.

Fonseca et al. (1982) describe a low cost planting method for transplanting *Zostera* shoots in some detail. Vegetative material is collected with a shovel. They suggest collecting from higher current areas from which, it has been shown, transplant material will have better growth rates and higher rhizome mat integrity. Clumps of shoots are pulled from the mats, which have been cleaned of sediment, and attached to a 20 cm piece of sturdy wire (e.g., coat hanger) bent into an "L" shape. A piece of construction paper is wrapped around the bundle that is then secured with a twist-tie. These planting units are then buried into the sediment so that the top of the anchor is covered. Fonseca et al. (1982) provide detailed man-hour estimates for this method. This is essentially the same approach as described by Thayer et al. (1985), who also provide information on optimum planting times for eelgrass on the east coast.

Curtis (1991) also finds the bare shoot approach effective but finds the anchor used inadequate in high current areas and a liability to swimmers. Instead, he ties a bundle of shoots to a flat wooden stick with cotton string. The planting unit is pushed into the sediment with the stick laid over the rhizomes and buried. He finds this method to be very successful with only a few failures that can be accounted for as poor site location (Curtis, 1991).

Halodule and *Syringodium* are sometimes observed to growth lengths of rhizome with shoots into the water column, referred to by some authors as "aerial runners." These may be collected and used as transplant material in place of digging up material. They provide the advantage that their collection is not disruptive to the donor grass bed, so their use is to be sought when they are available. Derrenbaker and Lewis (1982) used *Halodule* runners anchored to the sediment with staples to initiate restoration of a dredge and fill area in Florida. Within seven months, the transplanted area was nearly covered with *Halodule.* Thorhaug (1983) attempted a similar restoration that yielded a 31% cover in 10 months, interspersed with other colonizing species.

Fonseca et al. (1985) describe a detailed methodology for a low cost transplant technique for *Halodule* and *Syringodium.* The method has some similarities to that described above for *Zostera* (Fonseca et al., 1982). "Aerial runners," where available, may be used in place of digging up mats of seagrass, as is required for *Zostera.* The anchor used is a U-shaped piece of sturdy wire 20 cm long (like an erosion fabric control pin). In low current areas, the anchor may be pushed in place over a group of rhizomes or runners to secure them to the surface. In higher current areas, a planting units is assembled by attaching the anchor to the rhizomes with a twist-tie. They provide a table for calculating appropriate planting densities for each species to achieve full coverage over any chosen period of time from 50 to 200 days.

In pilot tests in Biscayne Bay, Thorhaug (1985) found *Syringodium* sprigs had poor survival and recommended against their use in transplants. She found that *Thalassia* sprigs did very well in terms of survival, in both high and low energy regimes, while *Halodule* did well at medium energy regimes but not high or low. In a large study (Thorhaug, 1987), she established that *Halodule* and *Syringodium* sprigs should not be planted in the winter. She was able to achieve very good survival of *Halodule* sprigs without anchors but lost much of it to winter storms, except in protected areas. Another planting, of *Halodule* and *Thalassia* sprigs, without anchors, showed very high survival of *Thalassia* after one year. *Halodule* was able to coalesce rapidly due to its rapid growth, but had relatively poor survival per transplant.

Fonseca et al. (1987a) describe in some detail what criteria should be set for good sprig quality to expect reliable transplants. They recommend that *Halodule* and *Syringodium* be transplanted first and allowed to coalesce before planting *Thalassia*. *Thalassia* should be transferred using sprigs only if there are no seedlings available, since sprig collection will lead to donor grassbed injuries. *Thalassia* sprigs are planted much the same as described for *Halodule* and *Syringodium* (Fonseca, 1985), but the sprig attached to its anchor is buried to the same depth from which it was harvested.

<u>Seeds or Seedlings</u>

Fonseca (1992) states that seeding of eelgrass in Chesapeake Bay has been reported to be successful, but Thayer et al. (1985) consider seeding eelgrass to be highly variable and not an option. In fact, only *Thalassia* has a record of successful seeding in the field. Thorhaug and Austin (1976) list the following advantages of seeding of *Thalassia*: revegetation by seeds is faster because of the rapid lateral expansion of the rhizomes, collection of seeds requires little or no injury to the donor grassbed, seeds are easy to transport, it is less expensive to seed than other transplant methods, and in practice, seeding is less depth-limited than turfing or plugging.

Thorhaug (1974) was the first to successfully establish *Thalassia* using seeds or seedlings. Seeds were removed from SCUBA-collected fruits. Seeds were held in running seawater following collection during which time seeds germinated into seedlings. These were secured to 12 cm plastic anchors and planted into an area previously denuded by a now-diverted power plant thermal plume in the Turkey Point area of Biscayne Bay. After 9 months, 70% of the plants had survived and were growing in place. The chosen planting area was considered ideal. It is a low energy area with a peaty substrate that provides a good attachment for roots. The area was covered with a "moderately dense" growth of *Thalassia* in two and half years after planting (Thorhaug and Austin, 1976). A later planting study in a more stressed region of Biscayne Bay demonstrated that seedlings could start growth as readily as they had at Turkey Point, but that after six months growth was less vigorous, suggesting limitations due to sediment or water quality. *Thalassia* seedlings grew better on beds of *Halodule* within this area than they did on bare sand.

Derrenbacker and Lewis (1982) hand-broadcast *Thalassia* seedlings over the transplanted *Halodule* bed discussed previously. This was an attempt to accelerate the natural successional process of *Halodule* to *Thalassia*. In a follow up study 7 months later, half the seedlings had survived, but no more information is available on this study. Thorhaug (1983) used the same technique of planting *Thalassia* seedlings over *Halodule* but achieved only 2 1/2% survival of the seedlings 10 months later. She also found in a pilot study for a larger planting in Biscayne Bay that while *Thalassia* seeds did well once established, there was a problem stabilizing newly planted seeds (Thorhaug, 1985).

Despite the rather mixed history of seed propagation of *Thalassia*, Fonseca et al. (1987a) concluded that whenever its seeds are available, seeding is to be preferred to plugging or springing for all the same reasons discussed above.

Summary of Methods

Thorhaug (1986) reviewed the published history of seagrass restoration and found that 21 groups had made 165 attempts at restoration worldwide, of which 75 had been successful. Thorhaug (1986) developed the following comparison.

Comparison between seagrass techniques.

	Plugs	Seeds	Sprigs	Turfs
Cost	high	low	medium	medium
Flexibility of situation	high	medium	medium	medium
Mechanization	extraction	planting	planting	planting
Transport	costly, difficult	easy	medium	medium
Damage to donor bed	high	none	medium	high
Use in high exposure areas	high	anchored only	medium anchored only	medium anchored only
Potential for survival	high	high (*Thalassia* only)	medium	high
Season for planting	can occur all year in tropics and subtropics	season differs for species	arctic, temperate, subtropics, seasonal	can occur all year in tropics and subtropics
Total attempts	71	25	53	16
Successes	37	14	12 (some pending)	8

This comparison summarizes many of the features, positive and negative, of the methods that have been discussed above and gives some sense of what techniques have been used most. Thorhaug acknowledges that many of the studies reviewed gave insufficient information to understand what had been done.

3.2.3.2.2.4 Recovery of Seagrass Beds

The question of how long it takes a seagrass bed to recover from some injury may be addressed at several levels. Most obviously, and most often, evaluations are given on the return of the physical structure and appearance of the grass itself. This may be given as percent cover, shoots per square meter, or simply as a subjective impression of looking like an uninjured natural grassbed. This does not address the status of the whole community, which can be complex and diverse, or of the habitat functions, which include primary and secondary productivity, current and turbidity modifying processes, nutrient transformations, etc. We find some studies that describe recovery in terms of grass cover, a few that discuss the accompanying animal community (or its most obvious aspects), but very few that deal with function.

<u>Seagrass Recovery</u>

In a review of eelgrass ecology in the Pacific northwest, Phillips (1984) concluded that not enough work had been done to establish the rate at which the eelgrass community develops. Curtis (1991) states that in a properly restored eelgrass bed, by the end of the second year, grass density should equal that of the donor bed or natural beds nearby.

Thorhaug (1979b) states that five years after the restoration work with *Thalassia* at Turkey Point in Biscayne Bay, adjacent areas were receiving seedlings from the transplant beds which had reached natural levels of abundance and biomass. She estimates that natural recovery of a *Thalassia* bed might take more than twenty years (Thorhaug, 1986).

Fonseca et al. (1987a) provide tables for seagrass planting densities to achieve coverage over any selected time period. For *Halodule* and *Syringodium* the possible time range is 50 to 200 days. They do not imply that "coverage" means density equal to a natural bed. *Thalassia* coverage occurs on the order of years rather than months and the selectable time range for *Thalassia* coverage is one to three years. They indicate that data are not yet adequate for these estimates in the *Thalassia* table to be reliable. Fonseca (1992) declares that natural shoot levels may be achieved in a *Thalassia* restoration in Tampa Bay in 3.38 years.

<u>Faunal Recovery</u>

Most studies of community restoration have focused on the animal community, and typically on a small component of the animal community. McLaughlin and Thorhaug, 1979 studied fish (mostly juveniles and larvae) and shrimp in the Turkey Point *Thalassia* restoration four years after planting. There were not significant differences detected between the restored beds and nearby natural beds. (Species composition is not necessarily similar to the natural habitat, however.) Thorhaug (1981) found at another Biscayne Bay restoration that within weeks of planting, fish and invertebrates were moving into the bed and using the replacement *Thalassia* blades for habitat, attachment and laying eggs.

In a controlled study of animal (mostly infaunal) recruitment to transplanted *Zostera* beds Homziak et al. (1982) found that the density of shoots was an important factor regulating development of the community. Total numbers and numbers of species were significantly related to shoot density and approached an asymptote at about 300 shoots/m^2. Fonseca et al. (1990) observed a newly naturally-seeded *Zostera* bed and found that in six months (December to June) the new bed had 85% of the numbers of fish and 64% of the numbers of shrimp found in a natural bed. (Species composition is not necessarily similar to the natural habitat, however.) They conclude that this rapid repopulation of the animal community is consistent with the intuitive concept that the rate limiting factor for faunal development in an eelgrass bed is shoot abundance. Hoffman (1991) studied fish utilization of a transplanted eelgrass bed. Utilization was high at the first study in three months, and in a year the transplant bed was essentially the same as an adjacent natural bed for the parameters measured.

3.2.3.2.3 Seagrass Restoration and Recovery: Summary and Conclusions

The most likely impacts of an oil discharge affecting a seagrass bed are the loss of many of the animals in the grassbed community and possibly a temporary slowing of growth of the seagrass, or even loss of exposed blades, but not death of the entire plant. Under these circumstances, it will be appropriate to allow the grass bed to recover naturally, accompanied by a monitoring program to ensure that this recovery takes place in a timely manner and in a natural direction.

When the roots and rhizomes of the seagrass are killed as well, however, it will be necessary to actively restore the loss. First, the condition of the site should be evaluated to ensure that it is suitable for restoration. If, as has occurred elsewhere, the loss of seagrass is accompanied by dramatic changes such as erosion or increased turbidity, it must then be decided whether it is better to restore on-site or off-site. This also applies if there is significant sediment toxicity left by the discharge. (Toxicity may be assessed by bioassays, for example.)

The decision for the actual restoration action to apply under any given circumstance is the province of an experienced expert in seagrass restoration. While the recommendations of Fonseca et al. (1982; 1985; 1987a) for seed/seedling restoration of *Thalassia* and sprig restoration of the others would seem to be the best available methods in a very general sense, it is quite possible that conditions for these approaches will not be appropriate for a particular case. Only an expert on site can make such a determination.

There are many areas still needing research in seagrass bed restoration. We need more detailed synoptic studies of restorations to determine what accounts for the success or failure of various methods (Lewis and Phillips, 1981). We need more information on recovery rates of restored and natural seagrass communities both for the seagrass themselves as well as the accompanying faunal community (Thorhaug, 1986). Fonseca (1992) provides a longer list from which we have elected several goals: How do we define the functional restoration of a seagrass bed? We need more data on growth and coverage rates for the various species of seagrass. Transplant optimization techniques should be developed. We need to know more about the role of genetic diversity (Fonseca, 1992). Finally, we need to define success and provide measures that can be readily used to evaluate effectiveness and success.

3.2.3.3 Freshwater Aquatic Beds (Submerged and Floating Vegetation)

No references either on impacts of oil discharges on freshwater aquatic beds or on restoration from such impacts were found. There is a paucity of information on restoration in these habitats from impacts of any kind. This may reflect a bias about the desirability of this habitat. In many instances, freshwater aquatic beds are viewed as nuisances. The U.S. Army Corps of Engineers has a research program for controlling aquatic plants (USACOE, 1992). Such nuisance beds may be largely the result of anthropogenic impacts such as eutrophication, or a bias toward anthropogenic uses (e.g., recreation) of a habitat over its potential natural values. While these values have not been elaborated anywhere in any detail, it is most probable that they are similar to those for nearby emergent habitats or for comparable marine habitats. Thus it should be expected that freshwater beds provide habitat for fish and invertebrate species, food for birds and other fauna, bottom stabilization and shorelined protection, reduce currents and alter sedimentation patterns, and roles in the nutrient cycles of the broader environment of which they are a part (Levine and Willard, 1990).

Levine and Willard (1990) give some very broad design guidelines for creation and restoration of fringe wetlands but are primarily concerned with emergent habitats. They provide a brief description of the Lake Puckaway, Wisconsin project to restore natural vegetation and gamefish and provide a food source for ducks. The project included the exclusion of carp through a series of measures, the establishment of a wave barrier, and the planting of wild celery, wild rice and sago pondweed. A variety of planting methods were used with mixed success. All the wild rice stands were lost. The wave barrier was removed after three years at which time the wild celery was able to provide the same function. The project was declared successful based on vegetation, water clarity and fish species (Levine and willard, 1990).

Wein et al. (1987) describe a habitat creation effort in mitigation of thermal pollution in a South Carolina lake. The restoration consisted of transplanting 100,000 plants, 30% of them submerged or floating vegetation, with the goal of accelerating the development of a natural balanced biological community. A nearby pond served as the model for the restoration as well as a source for transplant material. Problems encountered include water level fluctuations, selection of optimum planting times, and feeding on the transplant material by fauna. Transplants were reported to be growing and reproducing, but it was premature at the time of publication to declare the project a success (Wein et al., 1987).

Clearly a great deal remains to be learned about restoration of freshwater aquatic beds. Studies of their functional significance within the ecosystem (physical, chemical, and biological) would be useful in directing restoration efforts toward appropriate standards of success. The two restoration projects described above suggest that restoration of this habitat is still highly experimental. Information on optimum planting strategies and on the cultural needs for the various species involved will be important in increasing the reliability of this technology.

3.2.4 Mollusc Reefs

3.2.4.1 Review of Available Literature

Oyster reefs differ from the other biologically-defined structured habitats discussed here (e.g., seagrass or kelp beds) in that the community persists overwhelmingly on energy inputs from outside the community and dispenses wastes to the outside environment, rather than constituting an internally-productive complex system that recycles a large portion of its production and wastes. Furthermore, while it is indeed a community, with numerous species living in close proximity, there is less evidence that the oyster reef provides a wide variety of services or acts as an important structured habitat for other commercially or recreationally important species (Zimmerman et al., 1989). This considerably simplifies the question of restoration in that it largely reduces to a matter of the growth and biomass of a single species. Mussels may also form compact reef-like assemblages with properties similar to those discussed for the oyster reef.

The literature appears to be nearly devoid of any references to the impact of oil discharges on oyster reefs and no record of restoration of oyster reefs in response to oil discharge injuries was found. Neff et al. (1982) studied two populations of oysters impacted by the *Amoco Cadiz* oil discharge. Little growth occurred in these oysters for a year after the discharge, then growth returned to normal. Oyster tissues continued to be contaminated with petroleum hydrocarbons for 27 months after the discharge. This contamination apparently arose from oil leaching out of the heavily contaminated sediments.

Chan (1975) observed the impacts of an oil discharge on a large intertidal mussel bed (*Mytilus californianus*) resulting from the 1971 San Francisco oil discharge. Despite heavy oiling, mortality was very low. Two and a half years later the mussels were observed to be in a healthy state of recruitment with greater than pre-incident densities. Mussels (*Mytilus edulis*) exposed to an experimental oil discharge in Maine (Gilfillan et al., 1986) showed only a transitory elevation in tissue hydrocarbons (less than one month) and alterations of enzyme activity levels that lasted at least a few months, but no measurable impact on scope-for-growth (a laboratory measure of growth potential). Two years after the *Exxon Valdez* discharge, intertidal mussels were still lower in density and biomass at oiled sites than at unoiled control sites (EVOS Trustees, 1992).

Oyster bed restorations have been undertaken in response to a variety of causes. The beds or the oysters have been injured or destroyed by hurricanes (Munden, 1974; Berrigan, 1990), catastrophic freshwater flows (Hofstetter, 1981; Marwitz and Bryan, 1990; Bowling, 1992), dredging (Visel, 1988), improper maintenance and management of commercially fished beds (Kennedy, 1991) or disease. In addition, there may be a lack of substrate in an area believed otherwise suitable for oyster growth (Webster and Meritt, 1988).

A suitable oyster growing ground requires a firm substrate and suitable sites for attachment of oysters. A rocky bottom provides both of these, but it is difficult to harvest oysters from the rocks. A firm mud or sandy mud bottom provides a good substrate (Webster and Meritt, 1988; Munden, 1974), but a surface is needed to which oysters will attach, even if it is the oysters themselves. This may take several forms:

- Any of a variety of materials -- historically, oyster shells -- are planted in a thin layer on the firm substrate. When oysters in surrounding beds reproduce, the larvae settle on this "cultch" and grow on these surfaces;

- Seed oysters may be collected from areas unsuitable for growth and spread out on the presumably more suitable target bed. The source for these seed oysters is typically an intertidal bar or otherwise stressed area where oysters may be found growing under very overcrowded conditions and rarely reach marketable size (Berrigan, 1985); and

- Oysters may be "relayed" from areas that are closed to fishing due to bacterial contamination (e.g., sewage) to areas where they may grow out in clean water to marketable size for harvesting (Berrigan, 1985).

In the instance that no suitably firm substrate exists in an area believed to be suitable for oyster settling and growth, it is possible the ground may be stabilized. Webster and Meritt (1988) describe the methods for laying down a "foundation" in barren areas to allow its cultivation for oyster growth. Typically this consists of laying down a more or less thick layer of cultch material to solidify the bottom. Webster and Meritt (1988) provide a number of details that need to be considered, conditions that should be met, and costs associated with stabilizing oyster ground.

Often, a potential (or underproductive) oyster bed needs only fresh cultch to increase production. Under natural conditions, old oysters serve as cultch for new oysters. Traditionally, cultch laid by oystermen was the shells from shucked oysters of the region. It is not uncommon now, however, for those oysters to be shipped long distances and thus leave the system.

For a long time, dredged clam shell was a favored source of cultch in the Gulf of Mexico. It is cheap and provides a very good cultch medium. However, dredging was recently banned in Lake Pontchartrain, the major source of this shell, putting a premium on its use (Haywood and Soniat, 1992). There have been several recent efforts to look for alternative media for cultch in response to this change. Haven et al. (1987) found slate to be a poor substitute for oyster shell. Similarly, Mann et al. (1990) found that oyster shell was markedly preferable to expand shale or tire chips. Soniat et al. (1991) found that oysters set on limestone preferably to clam shell. The limiting factor was the higher density of limestone limiting its use in softer substrates. Haywood and Soniat (1992) found that both limestone and stabilized gypsum attracted more spat (settled oysters) than clam shell. The advantage of the stabilized gypsum is that it appears to be a benign product that provides a use for gypsum, otherwise a waste product.

In proposing a new bed it is important to consider all the environmental variables that will determine habitat quality for the oysters at various times of the year. These include temperature, salinity, suspended sediments, dissolved oxygen, pH (Kennedy, 1991), various qualities of the sediment, and proximity to other oyster beds as a source of spat. Laying cultch must be timed to the reproductive cycle of the local oyster populations. If it is laid too early, it may be fouled by encrusting organisms and sediment. If it is laid too late, the peak setting time will be missed (Morales-Alamo and Mann, 1990).

A review of restoration efforts suggests that under ideal conditions (a clean environment in an area conducive to oyster growth), an oyster bed may be largely restored to commercial utility in 1 to 2 1/2 years (Berrigan, 1990; Hoffstetter, 1981; Munden, 1974; Visel, 1988). This does not take account of the confounding effect that a severe oil discharge would cause, with its attendant contamination and possibly disruptive cleanup efforts.

3.2.4.2 Mollusc Reef Restoration and Recovery: Summary and Conclusions

An impacted oyster reef or bed should be restored to its original condition whenever possible. It is unlikely that subtidal oyster beds will need more than a brief period of depuration to return to pres-incident condition. Intertidal populations, however, might be more severely affected. Such populations are routinely found where environmental conditions are conducive to their growth (see Bahr and Lanier, 1981) and seeking an alternate site is likely to reduce the probability of a successful restoration. The most likely sites for off-site replacement may be the restoration of old oyster beds that may be underproductive and that can be helped through the addition of cultch, seed oysters, or perhaps relayed oysters.

Research is still needed on optimum placement of oyster beds and cultch, as well as on why some areas are especially conducive to settling spat while others are especially productive (Kennedy, 1991).

3.2.5 Coral Reefs

3.2.5.1 Review of Available Literature

Coral reefs constitute rich, highly complex, diverse, and productive biotic assemblages commonly found in tropical and subtropical coastal areas of the world. A description of these systems as found in South Florida, their ecology, environment, community composition, and management, are in Jaap (ed., 1984).

This review found no examples of coral or coral reef restoration in response to oil discharge injury. There are several studies that have examined the impact of oil discharges on coral reefs and some observations on natural recovery from these incidents. There are no known such studies in which the whole community was examined. It is assumed that when the coral recovers, the community that it is a part of recovers with it. Submerged corals do not seem to be particularly susceptible to oil discharges. Thus, submerged coral patches in the area of the 1968 *Witwater* discharge in Panama (Rutzler and Sterrer, 1970) and the 1975 Florida Keys discharge (Chan, 1977) showed no detectable injury. In both cases, no dispersants were used and weather conditions were conducive to keeping the floating oil separate from the submerged corals. Both of these studies were largely qualitative and did not take into account possible physiological impacts that would not be visibly evident.

Reef flat corals had disappeared two months after the 1986 refinery discharge in Bahia las Minas, Panama, and many of the shallow water subtidal corals were dead or dying (Cubit et al., 1987). Few animals had returned to the reef flats a year later and there was a 45% loss of coral cover at the heavily-oiled shallow subtidal reefs. Loss of coral cover wasn't significant at the deeper sites (Jackson et al., 1989). Assessment of longer-term recovery has been confounded by catastrophic low tides in 1988, but coral cover was still much lower than at control sites two years after the discharge (Keller et al., 1991). Jackson et al. (1989) suggested that some of the coral injury may have been aggravated by the use of dispersant, although only a small amount was used.

Cubit and Connor (1993) observed that rates of recovery of the various reef flat organisms affected by the Bahia las Minas refinery discharge varied with several factors, including the organisms's inherent growth rate, its mode of regeneration or recruitment, the severity of injury from the oil discharge, the existence of refuges near the oiled area which could provide a source for propagules for recruitment, and competition from other species. The stony corals in the study area suffered nearly 100% mortality and the slowness of their recovery was due in large part to their reliance on the growth of fragments of colonies washing into the affected area from adjoining, less-affected areas.

Birkeland et al. (1976) performed experimental field studies of the effect of Bunker C and diesel fuel on various marine communities in Panama. Their most important observations regarding coral were that oil may impact them physiologically, reducing growth rate in visibly unaffected corals. Further, this effect is quite variable in space and time and with species, requiring rigorous controls to properly evaluate. Ballou et al. (1987) exposed corals to dispersed and undispersed oil in a field experiment. They observed a distinct decline in coral coverage, other measures of community structure and function, and growth rates during recovery at the dispersed oil site. The undispersed oiled site showed slight decreases in coral coverage, but not in other community parameters. There were no measured effects on growth rate of the recovering corals.

Fucik et al. (1984) in a review of oil discharge impacts on coral reefs, proposed that a general lack of apparent acute impacts of oil discharges on corals only indicates that we are looking at the wrong variable. There may be sublethal responses (e.g., growth rate) that are important to the health of the coral community. If injury to the community cannot be properly identified, recovery cannot be evaluated. They acknowledge that the complexity of the coral reef system is such that it is unlikely the state of the whole system can be fully quantified, so it is important at least to determine the patterns of recovery of its major structural elements, the hermatypic corals and coralline algae.

Aside from oil discharge injury there are a number of other possible sources of injury to coral reefs or coral reef systems for which restoration methods have been attempted or proposed. Maragos (1992) and Woodley and Clark (1989) review a variety of causes of injury and methods of rehabilitation. Woodley and Clark (1989) classify such methods as either passive rehabilitation, which is any of a variety of impact mitigation actions that allow the reef to recover naturally or active rehabilitation in which the various organisms making up the reef community are manipulated to accelerate "recovery of value." By recovery of value, they mean increase in coral cover or reef fish or decrease in free-living algae, which may compete with the corals.

The primary and most obvious measure of reef injury and recovery is coral cover. Increase in coral cover may be accomplished through clearing existing surfaces or providing additional surfaces on which coral settlement may take place. Maragos (1992) lists a variety of techniques for accomplishing this including artificial reef construction, revetments, or breakwaters, or cutting reef flat quarry holes -- a means of adding a third dimension to a hard two-dimensional reef flat. Each of these provide surfaces for new corals to attach as well as crevices on surfaces that may provide habitats for the numerous other reef-dwellers.

Where injury results from an oil discharge, it is probable that these surfaces already exist. If the injury is sufficiently profound that natural recovery is expected to be very slow, the appropriate restoration may be transplantation. This is a technique still in its infancy. Results to date, however, have been promising. Maragos (1974) attached pieces of transplanted coral to iron frames with insulated wire and compared the resulting coral growth with natural coral colonization on artificial surfaces. Results of this short (18 month) study were mixed. Generally, larger transplant specimens were more successful than smaller ones. Maragos (1974) also studied natural recovery in a variety of areas. He concluded that transplantation is not to be recommended where natural colonization is likely (near a good source of larvae or where substantial live coral remains) since it will only reduce the time of recovery a few years. Results in Shinn (1976) tend to support this conclusion. Coral reefs that underwent devastating hurricane injury were able to recover so rapidly that the injury was undetectable five years later. Most of the fragments left unburied by the hurricane retained live coral such that the fragmentation in effect increased the number of growing centers. The staghorn coral that made up the greater part of this reef is a very rapidly-growing species. This example is given in support of Maragos' observations about not transplanting where source material already exists. There is no evidence in Shinn's study that the reef in question was in fact fully recovered. There is little information on the full diversity of corals nor any of the other species that constitute part of the reef system. It is quite possible that the reef in Shinn's study never reaches a high degree of complexity because of the high frequency of storm damage here. Griggs and Maragos (1974) observed that coral reefs in exposed areas are regularly disrupted keeping them in pioneer stages of succession whereas reefs in more protected areas may be more fully developed. Pearson (1981), too, has observed that reefs may be locally adapted to the periodicity of major storm events.

Hudson and Diaz (1988) performed pilot tests with coral transplanting at the site of a major ship grounding. The M/V *Wellwood* ran aground on Molasses Reef in the Key Largo National Marine Sanctuary, causing extensive injury to over 1000 m^2 of reef. Underwater cement was used to attach coral transplants (hard and soft corals) to the substrate as well as to reattach massive corals and repair fractures in the underlying reef framework. All the hard corals were still alive four years later, though the soft corals experienced considerable losses due to a storm.

Gittings and Bright (1990) have also studied the injuries to coral reef resulting from the M/V *Wellwood* grounding and have followed natural recovery over the ensuring five years. They found that recruitment has been dominated by species that brood larvae that then colonize near the parent colony. These typically are the small, abundant species. They concluded that transplantation could help the recovery of the larger massive corals. Typically, these larger corals broadcast their gametes to be fertilized in the water column. Recruitment of these species relies more on chance, and they are slow-growing. They conclude that in designing a transplant program, consideration should be given to the coral species' reproductive strategies. An additional benefit of transplanting massive corals is that they add to the structure of the environment, accelerating the recovery of other species (fish and invertebrates) that rely on surfaces and crevices as part of their habitats (Gittings and Bright, 1990).

Other methods of increasing coral populations might be to decrease mortality either by controlling disease or controlling predators (Woodley and Clark, 1989). These techniques remain experimental. A similarly untried but potential means of encouraging coral growth is through controlling growth of macroalgae that will compete with the coral for light and space. This may be accomplished through physical removal or by encouraging grazers (Woodley and Clark, 1989).

The other species that make up a coral reef community, especially invertebrates and fish, may also require augmentation to accelerate recovery of the reef. Mariculture and stocking of these species has been proposed as a possible future solution (Maragos, 1992), but techniques for this are not really developed. An alternative may be to replant seagrass beds and mangrove fringe, where lacking, which normally occupy or fringe the adjacent reef flat of many reefs in order to provide habitat for alternative life stages of some of the reef dwellers (Maragos, 1992).

Clearly, recovery time will vary with the extent of injury. Estimates of the actual time involved appears to be rather speculative at this point. Fucik et al. (1984) suggest that a coral reef may recover from localized natural disturbances in less than ten years so long as the area remains essentially healthy. Recovery from heavy impacts might take ten to twenty years and even longer for more severe injury. Loya and Rinkevich observed coral recovery from injuries caused by catastrophic low tides. While a clean area was found to be "flourishing" after only three years in an area with chronic oil pollution, there was almost no coral recolonization ten years later after the injury. Other discussions of coral reef recovery make the point that a great deal remains to be learned about the processes of succession leading up to a healthy, mature coral reef environment (Johannes, 1970; Fucik et al., 1984).

3.2.5.2 Coral Reef Restoration and Recovery: Summary and Conclusions

Recovery of coral reefs from extensive injuries takes so long, and any active restoration option is potentially so expensive, that prevention of injury to coral reefs from discharged oil should be a high priority. While coral that does not contact oil appears uninjured by oil floating over it, there is evidence that there may be subtler effects, such as on growth rate (Birkeland et al., 1976). It is important to be aware of such effects in evaluating injuries from an oil discharge and post-discharge monitoring should monitor for such effects.

In the event of some coral death after a discharge that leaves significant areas of live coral, natural recovery is recommended in most instances. Monitoring efforts should carefully evaluate whether particular species may be missing that could be aided by transplants. Where injury is extensive, i.e., near 100% mortality, serious consideration should be given to a transplant program to accelerate recovery of the reef. This, of course must take place in a whole-system perspective. If, for example, adjacent seagrass beds are injured, there must be restoration efforts expended there as well to ensure the sediment-stabilizing and habitat values that they provide are available. Another element of the whole-system perspective relates to source material for transplants. There must be a proper evaluation of the impact to the donor system of removing the transplant material. Other plants and animals in the system may have to rely on natural migration for recovery. Techniques do not yet exist for most species to culture and restock them at the proper scale.

Transplanting is still a relatively new technique and additional pilot projects should be undertaken to expand our knowledge of this technique and its limitations. Maragos (1992) proposes that we also need more work in culturing techniques for corals, other invertebrates and fish, and research into optimal stocking practices.

3.2.6 Estuarine and Marine Intertidal Habitats

The intertidal zone is particularly vulnerable to the impacts of oiling and cleanup operations. Populations of algae (e.g., *Fucus*), barnacles, limpets, amphipods, isopods, molluscs and marine worms are affected.

However, the shoreline is also an area where natural processes rapidly remove oil following a discharge. Natural washing and abrasion caused by wave action and tidal flushing are effective in restoring the shoreline to its pre-incident condition. In Prince William Sound after the *Exxon Valdez* oil discharge, waves and twice-daily tides of 3 to 6 m moved sediment particles to abrade oil and wash it away. This oil was dispersed into the ocean and broken down by biological processes (Owens, 1991).

The rate of this natural cleaning occurs as a function of the wave action (energy) that reaches the shoreline, the thickness or depth to which oil has penetrated the substrate, and the mobility of beach sediments (Owens, 1991). Oil penetrating cobble beaches is removed by tidal flushing. Storm waves redistribute sediments across a beach and expose underlying oil. Natural microbes also work on the oil. Biodegradation is one of the key processes that remove oil.

For oils with high fractions of soluble and volatile components, the contamination will generally not remain on the shoreline long enough for restoration actions to be necessary. Indeed, Ganning et al. (1984) recommend that no action be taken in restoring rocky shores, beaches, and tidal flats following contamination from discharges of light refined petroleum products. The toxic components of these products are highly volatile and natural processes (including biodegradation) will remove the toxicity rapidly.

Cox and Cowell (1979) suggest that in most cases oiled shorelines are best left to recover naturally, as the disturbance of cleaning often causes more harm (ecologically) than the original oil contamination. They cite the *Amoco Cadiz* discharge as a case in point.

It has also been observed that *Fucus* survives oiling due to mucus cover, but is impacted by intrusive cleanup techniques (R.Hoff, NOAA-HMRAD, pers. comm.). Cox and Cowell (1979) also argue that shorelines are best repopulated naturally since biota are seeded planktonically. Recently, a study published by Foster et al. (1990) concluded that shoreline cleanup methods "appear to be much more damaging to shore life than the discharge itself". NOAA found "there is no net environmental benefit to be gained from shoreline excavation and washing," after examination of beaches cleaned following the *Exxon Valdez* oil discharge in Prince William Sound (Golob's Oil Pollution Bulletin, 1990a).

However, in some cases active restoration has been recommended, i.e., when contamination is heavy and long term (such as by crude oil or by insoluble, slowly degrading toxic substances). Cox and Cowell (1979) and Ganning et al. (1984) stress that only mechanical cleaning methods or low pressure cold-water washing be used in the case of heavy oil contamination. However, if the shoreline is valued for some public use (recreational or commercial), more drastic cleaning measures might be called for, such as steam cleaning (Cox and Cowell, 1979; Ganning et al., 1984) or washing and replacement of sand (Bocard et al., 1989).

Recently, considerable research has focused on bioremediation for restoration of oiled shorelines (Office of Technology Assessment, 1991; Hoff, 1992). The objective of bioremediation is to accelerate the natural biodegradation process by the addition of microbial cultures to boost natural populations of hydrocarbon-degrading bacteria and oleophilic ("oil-loving") formulations and/or fertilizers to stimulate natural bacterial breakdown of petroleum hydrocarbons. Since hydrocarbons are a source of energy (reduced carbon) but have low nitrogen and phosphorus content, fertilization can supply these nutrients that may limit bacterial growth rates (Halmo, 1985). Other potentially limiting factors are oxygen and temperature. Oleophilic formulations (surfactants) address the problem that oil is as a whole non-polar and does not mix with water easily. Surfactants are often added with the fertilizer to increase the binding to the oil, as well as break up the oil which facilitates physical removal and increases surface area available to microbes and to oxygeneration (Sveum, 1987). Field tests of this methodology have shown success, but it is not clear whether the increase in the disappearance rate of the oil was due to stimulated biodegradation or to the surfactants that increase physical removal rates (Halmo, 1985; Sveum, 1987; Sveum and Ladousse, 1989; Kremer, 1990; Golob's Oil Pollution Bulletin, 1990b; Hoff, 1992). Hoff (1992) provides a concise summary of bioremediation research on effectiveness to date. Her summary shows that bioremediation using fertilizer and oleophilic agents (but not microbe additions) was partially effective on Prince William Sound beaches following *Exxon Valdez* oil discharge, and especially effective on subsurface oil in gravel beaches, but that other field studies following discharges are inconclusive. Tests of microbial additions have not proven effective in any case to date. The problem seems to be that the added microbial cultures are not adapted to the ambient conditions, and are out-competed by indigenous strains. Fertilizer and oleophilic additions do show promise for success and deserve further research. In any particular location, it needs to be determined what is limiting to hydrocarbon-degrading bacteria. Additives that remedy the limitation should prove effective.

3.2.6.1 Intertidal Rocky Shores

Numerous areas of the northeast and west coast of the U.S. and large areas of Alaska consist of rocky shoreline. Seventy-two percent of the shoreline affected by the *Exxon Valdez* discharge was bedrock. Oil coats the rock surfaces and tidal pools, and affects algae, molluscs, crustaceans and infauna that are resident to this type of habitat. The longevity of oil discharge-related injuries depends on the degree of wave activity (Gundlach and Hayes, 1978). In exposed areas, oil is removed rapidly, while in sheltered areas it persists for years.

A review of oil related literature indicates that hot-water washing, steam-cleaning, sand blasting, flushing (low pressure) ,and bioremediation techniques have been used on oiled rocky shores for response and restoration.

3.2.6.1.1 Case Studies of Oiling of Intertidal Rocky Shores

Seventy-two percent of the shorelines effected by the *Exxon Valdez* discharge in 1989 were rock outcrops and headlands separated by mixed-sediment beaches including boulders, cobble, and fine sediments. A number of natural processes worked to remove oil from these shorelines. Natural washing and abrasion caused by wave action and tidal flushing were the most important processes by which oil was removed from rocky shores in the months following the discharge (Owens, 1991; Michel and Hayes, 1993). Waves and tides moved sediment to abrade oil from rock and wash it away. The rate of such natural removal was a function of the intensity of wave action, thickness and depth of penetration of oil, and mobility of boulders and sediments.

Hot-water, high pressure washing was used on oiled rocky shorelines throughout Prince William Sound following the *Exxon Valdez* discharge and were shown to eliminate the majority of the flora and fauna from large areas of shoreline (NOAA, 1991). Hot-water washing involves the use of 60° C seawater at pressures of about 100 psi. In conjunction with the thermal stress, the pressure is sufficient to dislodge all but the most firmly attached barnacles and algae. Evidence of survival of these same taxa for several months on heavily oiled and untreated beaches clearly indicated that "there is no net environmental benefit to be gained from shoreline washing" (Golob's Oil Pollution Bulletin, 1990). In addition, this treatment has the potential of aggravating the injury to the rest of the environment caused by the oil discharge.

Studies were conducted in Prince William Sound in 1989 to determine the short-term impact to biota of hot water washing treatment. Additional surveys were conducted in 1990 to document recoveries of littoral habitats from the effects of oiling and subsurface treatment. Sampling focused on three intertidal habitat types of particular importance in Prince William Sound: protected rock, protected sand/gravel/cobble and exposed boulder/cobble. Three elevations of the intertidal area were surveyed. The use of high-pressure, heated water in rocky habitats resulted in significant effects on the intertidal flora and fauna of the area. Available data indicates that the 1990 condition of intertidal biota at many oiled areas would more closely resemble that at unoiled sites had shoreline treatments not been applied (Houghton et al., 1991a,b; 1993a,b). In areas cleaned most rigorously, complete loss of mussels and rockweed eliminated habitat for several species. Surveys in July of 1991 showed fewer statistically significant differences between biota of unoiled rocky shorelines and those of hot-water washed shores. However, full recovery is not expected for several years (Houghton et al., 1993a,b).

Bioremediation was attempted for remediation of oil-contaminated shorelines following the *Exxon Valdez* discharge. The fertilizer Inipol was used to stimulate the growth of naturally- occurring bacteria that degrade hydrocarbons (Crawford, 1990). However, the technique was not useful on rock shorelines contaminated by oil because fertilizers would not cling to vertical structures (Crawford, 1990).

Hot water washing has been observed to be more detrimental to intertidal biota than no action in other oil discharges. Broman et al. (1983) observed that hot water cleaning after an oil discharge in the Baltic Sea did more harm than good and slowed recovery dramatically.

In the *Torrey Canyon* discharge near Cornell, England, hot-water washes were implemented with a toxic dispersant (NOAA, 1991) in efforts to remove oil from the rocky shoreline. Following this incident the injuries were extensive. The dispersants were effective at reaching into crevices and tide pools, resulting in nearly complete mortality of fauna, and severe impacts to flora, over large areas. Southward and Southward (1978) observed that recolonization and recovery of rocky shores in Cornwall took 5-8 years if the shores were lightly oiled and received light dispersal treatment. Recovery took 9-10 years or more if the shore received repeated dispersant treatment. No sites were observed (or available) that were left untreated.

The February 1990 grounding of the *American Trader* off Huntington Beach, California oiled fourteen miles of southern California beach with Alaskan North Slope crude oil. From mid-February to mid-March the rocky shorelines that were affected were systematically cleaned using a variety of ambient-temperature, hot-water flushing, and spraying methods. The California Department of Fish and Game wardens set temperature constraints for each segment of rocky shoreline based on bioassays of marine life at each location (Card, 1991). Accurate assessment of the discharge and shoreline treatment impact cannot be made until data is released by the U.S. Fish and Wildlife Service and the California Department of Fish and Game.

The December 1988 discharge of 231,000 gallons of Bunker C fuel oil from the *Nestucca* oiled rocky intertidal shores of the outer coast of Washington state and Vancouver Island. Kinnetics Laboratories (1993) monitored recovery of intertidal biota after oiling, as compared to artificially cleared (i.e., scraped and burned) and control plots. Sampling included measurements of percent cover and abundance. After three years only two of five oiled plots had recovered (were not significantly different from the control plots). None of the cleared plots had recovered. *Fucus* spp. were nearly absent in oiled plots at the end of the study. Thus, full recovery from oiling is likely to be longer than three years, even where oiling is relatively light (as in the *Nestucca* case).

On March 16, 1978, the *Amoco Cadiz* grounded on the Coast of Brittany, France, and discharged its entire cargo of 223,000 tons of light crude oil. Shoreline cleanup was primarily performed with mechanical methods. Oil degradation on rocky shores was reported to be complete in two years, with only slight traces of oil remaining (Seip, 1984). However, this report states recolonization was still lacking in exposed areas after five years. In sheltered bays only some species had repopulated the area.

The *Esso Bernicia* discharged 8,000 barrels of Bunker C oil in December 1978, north of Scotland in the Shetland Islands. The rocky shoreline was inhabited by typical intertidal communities: rockweed, barnacles, and snails. A massive response was mounted early in 1979 with manual bagging of oiled debris as the principal method. Dispersants were used extensively on the water. However, trial applications on the oiled rocky shoreline were ineffective (Rolan and Gallagher, 1991) in many areas, or limited, or no cleanup was attempted. While a few species recovered rapidly on the cleaned sites, most did not. Eight and nine years later, none of the cleaned sites had recovered (Rolan, 1991; Rolan and Gallagher, 1991). At the same time it was reported that no significant effects on the abundance of populations on the uncleaned shores could be attributed to the *Esso Bernicia* discharge.

Steam cleaning and sandblasting can also be used to remove oil from rock. These techniques use high-pressure jets of steam or sand to physically remove oil from the contaminated surface. The high temperature, high pressure streams can severely erode the sediment around the rock and injure any uncontaminated fauna or flora in the area. The review of oil-related restoration literature did not include the use of either of these procedures since they are clearly not advisable restoration techniques for reducing biological injuries. (However, in certain areas, aesthetic or other non-biological services may make these actions desirable to reduce natural resource damages as a whole, i.e., be of net benefit. See Section 5.)

3.2.6.1.2 Experimental Studies on Intertidal Rocky Shores

Oil discharge research has involved several experiments to evaluate the effects of oil on shorelines and the effectiveness of cleanup or restoration actions. Under controlled conditions, oil has been discharged on shorelines in field studies. Laboratory and wave tank experiments have also been conducted and considerable knowledge has been obtained. Most field experiments were performed outside of North America.

Broman and his associates used Russian crude oil in an experimental discharge on exposed Baltic rocky shores dominated by lichens and algae. Water at 90°C and 2100 psi was efficient in freeing oil, but vegetation was dramatically reduced (Baker et al., 1993). Mussels placed in net bags offshore from the site showed significantly higher hydrocarbon levels in their tissue following this hot water washing.

A sheltered rocky shore in the United Kingdom dominated by brown algae was oiled and the algae cut. This removal allowed for colonization by the algae opportunist, *Enteromorpha* and the decrease in fauna due to lack of habitat structure (Baker et al., 1993). Brown algae is relatively resistant to oil and is slow growing. Removal was not an effective method for restoring the habitat.

The same literature review by Baker et al. (1993) details several experimental discharges where dispersants were employed in rocky shoreline habitats. Evidence showed that some oil/dispersant treatments are more injurious than oil alone. Considering the efficiency of natural cleaning that has been documented for exposed rocky shores, the use of dispersants would not be recommended. In sheltered areas dominated by algae the question is more complex. If invertebrates are killed by oiling, the use of dispersants has been shown to speed up recolonization.

3.2.6.1.3 Intertidal Rocky Shore Restoration and Recovery: Summary and Conclusions

Several major oil discharges have impacted rocky shorelines over the past ten to fifteen years. Only in Prince William Sound, following the *Exxon Valdez* discharge, were various cleanup and restoration techniques systematically studied for rocky shorelines. The consensus of most biologists is that most shoreline treatments do more harm than good to intertidal resources and may delay environmental recovery (Houghton et al., 1991a,b; 1993a,b).

On high energy, exposed rocky shorelines, wave action removes essentially all oil within weeks (Gundlach and Hayes, 1978). On sheltered rocky coasts, oil may persist for years depending on wave action and degree of oiling. Experience has shown that natural recovery is the least disruptive to native fauna and flora and allows for the shortest period of recovery. Bioremediation shows promise in aiding this recovery, but requires further study to determine effectiveness under a variety of conditions. Although hot water washing, steam cleaning, and sand blasting have been used to remove oil from rocky shorelines, none of these techniques has aided in the recovery time for the habitat or its associated marine life. Where oil removal is desirable to reduce sources of contamination and improve recovery of non-biological services, low temperature and pressure flushing is successful with several types of (lighter) oil and does not further injure biological habitats.

The time necessary for recovery is dependent on many environmental conditions such as temperature and wave action, and oil discharge characteristics. Baker et al. (1990) reported that rocky shores in the Baltic Sea had nearly recovered by one year after the *Tsesis* discharge of 1977. As cited by Ganning et al. (1984), recovery from a medium fuel oil discharge in the Baltic Sea followed by mechanical cleaning took four years, recovery from a Bunker C discharge in Nova Scotia took greater than six years and recovery from a No. 2 fuel oil discharge in Baja California took over ten years. In contrast, Keller and Jackson (1991) summarize recovery of intertidal rock reefs in Panama following a medium crude oil discharge as complete by one year. In general, natural biological recovery time for exposed rocky shoreline is about five years and about ten years for sheltered rocky shoreline (Booth et al., 1991). These are broad generalizations, but consistent with field studies.

Many environmental indicators are used to evaluate the recovery of oiled habitats. Measurements include physical and chemical evaluations of the amount of remaining oil. In vegetated habitats measurements of the size, densities and distributions of the key plant species should be made. In all habitats, measurement and evaluation of community structure, population characteristics and adverse effects on individual organisms are appropriate (Booth, 1991). Species abundance and biomass are most commonly measured. Section 3.2.10 provides further discussion of monitoring considerations for intertidal habitats.

For rocky shorelines, the upper, middle, and lower intertidal elevations need to be evaluated separately due to different community structures and interaction with tidal cycles. Sampling and evaluation should occur in each season throughout the monitoring program. Rocky shorelines need to be monitored for five years in exposed areas, ten years in sheltered areas from the time of injury in the case of natural recovery or from the time response and restoration actions are completed.

3.2.6.2 Intertidal Cobble-Gravel Beaches

Several major discharges in recent years have occurred along course-grained shorelines that contain extensive cobble-gravel beaches. These include the *Metula* (1974), *Amoco Cadiz* (1989), and *Exxon Valdez* (1989). Deep penetration and burial of oil is common on gravel beaches affected by a discharge, creating the potential for oil to remain for several years. In sheltered areas, heavily oiled beaches may convert to gravel pavements.

Medium-pressure flushing, sediment washing, sediment agitation, berm relocation, and bioremediation have been tried to restore cobble-gravel beaches affected by oil discharges. Below is a review of the available literature documenting these actions.

3.2.6.2.1 Case Studies of Oiling of Intertidal Cobble-Gravel Beaches

Gravel is the most common sediment type found on beaches in the Prince William Sound area. Several response and restoration actions were studied on this habitat type following the *Exxon Valdez* discharge in March 1989. Although cleaning of rock-cobble shorelines following other oil discharges has been performed, the literature does not contain scientific data on subsequent recovery. Thus, review of case studies is focused on the *Exxon Valdez* case.

Hot-water, high-pressure washing had the same effect on biological communities in this habitat as was seen with rocky shorelines, i.e., near complete mortality of fauna and flora (see Section 3.2.6.1.1).

Washing and flooding of cobble-gravel beaches was effective in floating oil to the surface and transporting it down slope for collection. However, intertidal habitat may be physically disturbed as sand and gravel are mixed and transported. This sediment may travel into the subtidal area and bury benthic organisms.

Grim Beach, mainly composed of gravel and angular cobble with a subtidal zone of sand, was extensively studied following the *Exxon Valdez* discharge. This area was covered with moderate to heavy concentrations of oil in the spring of 1989 and received more treatment than any beach on the outer coast (Dudiak and Middleton, 1991). Initially, Grim Beach was hot-water washed. Additional treatment included manual cleaning, mechanical working, and bioremediation. It was impossible to separate the effects of oil from the effects of initial hot water treatment on the biota of Grim Beach. No pre-incident data was known for the site. All the taxa that were abundant at the reference site at One Haul Bay and most other sites on the outer coast in 1990 were not seen at Grim Beach even by 1991. These reference sites were oiled. However, their cleanup included manual cleaning and/or bioremediation, but not hot-water washing. Large amounts of bioremediation materials were used on Grim Beach and were apparently very detrimental to biota. This site is used as an example of the difficulty in isolating the effects of treatments. Often, many technologies were used at the same location and decisions were changed during the period of response and restoration.

A Rockwash was developed by the Homer Area Recovery Coalition and used to clean Mars Cove. This portable machine was designed to remove gross oil product from beach rocks and gravel. It is a mobile, self-contained recirculating wash system which employs a dual stage filtration and pumping system that cleans and recirculates wash water. After washing is complete, the rock and gravel are returned to the beach. Although hot water and agitation are employed which would destroy species which adhere to these rocks, no additional injury is done to the habitat. Oil is removed, not forced further into the substrate. No recovery estimates or studies were attempted.

In Prince William Sound, wave action over the 1989-1990 winter months, along with biodegradation, considerably reduced the amount of oil on cobble-gravel shorelines. In 1990 Exxon continued using bioremediation and other non-intrusive techniques which would not interrupt biological recovery (Owens, 1991). A report issued by USEPA, Alaska Department of Environmental Conservation (ADEC) and Exxon estimates that bioremediation accelerated natural biodegradation five fold and occasionally as much as tenfold (Prince, 1990).

Part of the oil which remained after the winter was located on the highest parts of the beach. Oil had stranded on berms above the normal limit of wave action. A program was developed to relocate the oiled berm sediments back down to the beach to expose them to more effective bioremediation and natural cleaning by wave action. Berm relocation, as a method to accelerate subsurface oil removal, was carried out during the summers of 1990 and 1991. Relocation involved movement of oiled sediments from the inactive beach face into the upper intertidal zone where sediments could be cleansed by wave action. In 1992, a marked decrease in subsurface oil in the upper intertidal zone was observed a few months after berm relocation (Michel, 1993). Surveys showed the recovery time to be very site-specific. Oil was removed within months in some areas and not yet accomplished after one year in others. In planning such projects it is very useful to have detailed data on wave conditions, sediment types, longshore currents and seasonal storm patterns at a discharge site (Michel, 1993).

In some cases, following movement of berm material to the intertidal zone, fertilizers were added to aid biodegradation (Owens, 1991). A marked reduction in surface and subsurface oiling followed this treatment program. Thus, bioremediation of this type may be a useful restoration option.

3.2.6.2.2 Experimental Studies of Intertidal Cobble-Gravel Beaches

Experimental laboratory tests were conducted by Exxon to better understand the interactions between beach sediments in Prince William Sound, seawater and oil residue remaining on the shoreline following the *Exxon Valdez* discharge. A column flow apparatus was designed to simulate tidal flows. Sediments and rocks for the study were taken from Prince William Sound.

Briefly, the study concluded:

- Residue on rocks consists of a colloidal emulsion of oil, brine, and fine particulate matter;

- The emulsion does not adhere strongly to beach rocks;

- The high surface area of the oil/water interfaces in the emulsion should provide access to bacteria and, therefore, increase biodegradation; and

- Because the particles of the emulsion either float or are neutrally buoyant, they will be carried long distances once eroded.

This study helps to explain why natural oil removal was so extensive on most beaches during the winter following the *Exxon Valdez* discharge (Bragg et al., 1990).

Several experimental sediment washers have been tested but few have been used in actual discharge situations (Owens, 1992). The purpose of sediment washing is to remove oiled surface material, cleanse the sediment, and return it to the shoreline. The oiled substrate is removed using heavy equipment or hand tools and placed into a washing unit. Such units can be built for the purpose but are not commercially available. Portable or truck-mounted cement mixers can be adopted for this purpose. Washing solutions may include cold or hot water or a dispersant/ bleach cleaning agent solution. Sediment washing is primarily used on gravel, pebble or cobble shorelines where other cleanup techniques are often ineffective. It is only acceptable for low productivity areas since organisms that inhabit the sediment will likely be destroyed.

3.2.6.2.3 Intertidal Cobble-Gravel Beach Restoration and Recovery: Summary and Conclusions

The recommended alternatives and actions for restoration of cobble-gravel beaches depends on the relative significance of biological versus non-biological services affected. Where non-biological services (beach use, aesthetics, etc.) are more important (of higher value), cleaning of oil on and in the beach may be desirable. Even where biological services are the only values, the long-term continuing source of contamination from a cobble-gravel beach may be of concern enough to warrant its removal. However, many of the cleansing techniques are injurious to beach biota. Thus, the least injurious actions should be considered first.

Bioremediation has shown promise in low energy areas, and under certain conditions. If nutrients are limiting biodegradation then fertilizer application may enhance recovery.

Low pressure flushing with ambient temperature water is preferable over more drastic washing actions. Hot water washing should only be used where non-biological services are highly valued and overweigh the total loss of biota caused by the action.

Oil that came ashore in Prince William Sound on cobble/gravel beaches generally stranded onto the upper third of the intertidal zone. Most of this surface oil was naturally removed during the first winter storm period by wave action. This oil was completely gone in August 1992 surveys (Michel, 1993). During these same surveys, beaches classified as cobble/boulder with berms retained significant subsurface oil. The reason for persistence of this oil is the well-developed armor of cobble that is formed over the fine-grained subsurface sediments. This armor shields underlying sediment (and oil) and is moved only during major storms.

Berm relocations were carried out in 1990 and 1991. Oiled sediments were moved into the upper intertidal zone where they could be naturally cleaned by wave action. All the berm-relocation studies showed a marked decrease in subsurface oil in a period of a few months after relocation. However, details on wave conditions, sediment types, currents, and seasonal storm patterns determine cleansing rates for each site.

Although many cobble-gravel shorelines have been affected by oil discharges, documentation of (biological) recovery time is not available. Booth et al. (1991) estimates recovery times of less than one to ten years depending on shoreline exposure, similar to the case with rocky coast. The maximum natural recovery time for exposed beaches is estimated at five years, ten years for sheltered beaches.

Monitoring programs for cobble-gravel beaches should consider the factors discussed for evaluating rocky shoreline recovery, as well as the general considerations in Section 3.2.10. This habitat should be monitored for a ten year period on a seasonal basis following injury and the completion of restoration actions.

3.2.6.3 Intertidal Sand Beaches

Sand beaches may be cleaned up and/or restored following an oil discharge by flushing, sediment agitation, sediment washing, substrate removal, use of a beach cleaning machine, incineration, and with the use of bioremediation techniques.

3.2.6.3.1 Case Histories of Oiling of Intertidal Sand Beaches

On December 21, 1985, the *Arco Anchorage* ran aground in Port Angeles Harbor, Washington discharging 239,000 gallons of Alaska North Slope crude oil. Oil percolated into beach sediments on Eliz Hook, the most heavily oiled area. It was determined that large enough quantities of oil were trapped in the sediment to warrant removal. A removal method incorporating physical agitation to a depth of 12 inches and high pressure water jets was used to effectively remove entrained oil (Levine, 1987). Chemical analyses of beach sediments before and after the agitation program indicated that the method was very successful is removing oil. More than 74 percent of the crude oil was removed from areas of heavy beach contamination (Miller, 1987). Biological recovery data are not available.

Following the *Amoco Cadiz* discharge in 1978, sampling of the sandy beaches on the northern Brittany coast was conducted. No restoration actions were noted. A period of "degradation" and "impoverishment" of the fauna lasted two to three years followed by a "recovery" of the original fauna. Microfauna had returned to normal by 1983, five years after the discharge (Bodin, 1988).

The *American Trader* discharge in February, 1990 occurred off Huntington Beach, California and resulted in 9,500 barrels of Alaskan North Slope crude oil being released. Extensive cleanup was performed (but no restoration to date, although the damage assessment is on-going and restoration is being planned). Since the beaches were major recreational areas and low-profile shorelines subject to constant erosion, all oil removal was performed while minimizing sand removal, sand compaction, and other impacts to the environment. Workers shoveled oil sludge and contaminated sand into plastic bags that were removed to a landfill. Follow-up recovery studies of biota are not available.

Nutrient-enhanced bioremediation was tested at several locations in Prince William Sound following the *Exxon Valdez* discharge. These sites included sand, gravel, and cobble beaches. Visual observations suggest enhanced biodegradation occurred on the beaches treated with Inipol, which was applied in slow-release briquettes and dissolved solutions of inorganic nutrients. Samples of oil from fertilizer-treated beaches, taken at the same time as oil was visually disappearing, showed substantial change in hydrocarbon composition which indicated extensive biodegradation (Glaser, 1991). Recovery of infauna was not measured.

A mobile sand-washing plant was used following the *Amozzone* fuel oil discharge in 1988, but as with most previously-noted case histories, no ecological studies were conducted. However, it was concluded that washing oiled sand facilitates natural sediment decontamination by making sediment more mobile under tidal action, accelerating the recolonization process (Booth et al., 1991).

Another method of cleaning (as restoration) that has been tried with some success is beach agitation, which allows oil trapped in the beach to evaporate and degrade more rapidly (Miller, 1987). This restoration action was used on a heavily-oiled Rhode Island beach, which is highly valued for recreation (French et al., 1990), following the *World Prodigy* oil discharge in June 1989. This method is less harmful ecologically than sand washing, but is, of course, less efficient at removing the contamination. Blaylock and Houghton (1989) suggested that beach agitation after oiling appears to improve recovery rate of benthos, but did not provide estimates of time required.

Keller and Jackson (1991) summarized recovery of sand beaches in Panama following oiling as being complete by one year, except for certain species. Bodin (1988) observed recovery of three sand beaches in Brittany, France after the *Amoco Cadiz* oil discharge over the years 1978 to 1984. Recovery of the meiofauna was complete by 1983 (five years). Baker et al. (1990) cite evidence from the Baltic Sea after a 1970 discharge of medium and heavy fuel oil with mechanical cleanup, where recovery took four years. Judd et al. (1991) observed that Texas dune vegetation took 2-3 years to recover from removal experiments.

3.2.6.3.2 Experimental Studies on Intertidal Sand Beaches

No documentation of experimental studies evaluating effectiveness of restoration alternatives and actions were found in the literature.

3.2.6.3.3 Intertidal Sand Beach Restoration and Recovery: Summary and Conclusions

Exposed beaches will recover following natural cleaning from waves and wind. Thus, low wind and wave environments of sheltered beaches will require a greater period of time for natural recovery. Bioremediation, beach agitation, and low-pressure flushing may assist in removal of oil and hasten recovery.

Since sand beaches are characterized by highly mobile sediments, low-pressure flushing and agitation actions may not necessarily be lethal to biota. However, quantitative documentation of this is lacking. More disruptive actions such as sediment washing, sediment removal and replacement, and incineration will certainly be lethal. Thus, these latter actions should only be considered where non-biological services are more important (of higher value) than biological services of the biota present and surviving the discharge.

Statistical analyses of changes in oil residues on beaches in Prince William Sound demonstrated that bioremediation was successful in accelerating oil removal. Results of a joint USEPA, the state of Alaska and Exxon study show that on fertilized beaches the rate of oil biodegradation was from three to five times faster than on adjacent, unfertilized control beaches (Bragg et al., 1993).

Exposed beaches, subject to wind and waves, are dynamic habitats characterized by low biological diversity. Recovery would be expected to occur within a five-year period. The stable environment of a low energy, sheltered beach can sustain diverse communities and will likely require up to ten years for recovery.

A monitoring program should continue seasonally throughout the expected recovery period and consider the points noted on rocky shoreline monitoring, as well as general points in Section 3.2.10.

3.2.6.4 Intertidal Mud Flat

3.2.6.4.1 Case Studies of Oiling of Intertidal Mud Flats

Oil penetration is minimal in mud flats because sediments are fine and oil is usually lifted from the mud by standing water and rising tides. However, mixing into the mud might occur in storms or high current velocities. Bioturbation will also work oil into the sediment if it remains for any length of time. A review of the literature did not locate case studies on impacts or recovery rates.

3.2.6.4.2 Experimental Studies on Intertidal Mud Flats

Two series of experimental trials have been conducted in the United Kingdom to access cleanup and restoration actions for mud flats affected by oil. However, in neither case were biological impacts monitored. Presumably removal of oil would improve recovery, but it remains undocumented.

Experiments at Stert Flats included testing of flushing, skimming, scraping, and the use of absorbents. Low-pressure flushing was found difficult to implement on mud flats because of problems in obtaining enough water for flushing and in collecting the oil removed. Low-pressure flushing techniques proved useful for soft sediment cleanup if there is a readily available source of water, if pumps can operate on the flat without sinking into the mud, and if there is a means of collecting the flushed oil. Transportation of equipment and personnel must also be conducted over the soft mud, requiring hovercraft or other amphibious vehicles. To minimize the amount of water used, an additional experiment was conducted in which flushing water was recycled (Abbott et al., 1993). Straw matting can be used as a sorbent for removing oil emulsion from the surface of mud flats, although significant amounts of mud are also removed. A straw matting boom did prove successful in protecting a salt marsh. These experiments are continuing.

Field experiments involving low-pressure flushing were carried out with 85 percent efficiency in clearing fuel oil mousse from sheltered center-tidal sand/mud flats (Baker et al., 1993). The technique raised the water table and distributed the surface sediments sufficiently to liberate oil that had penetrated the mud. It would be effective with thick, firm sediments. Use of flowing water was also found to protect mud flat surfaces.

3.2.6.4.3 Intertidal Mud Flat Restoration and Recovery: Summary and Conclusions

Little work has been done to study restoration of mud flats following discharges of oil. Flushing has been shown to be effective at removing oil under certain conditions, but for most locations it is logistically difficult.

Residence time for oil discharged onto sediments of mud flats is relatively short because of physical removal by tides, low affinity of hydrocarbons to wet substrate, and low sediment permeability. Response or restoration actions may cause additional injury, primarily by forcing oil into mud when equipment and/or personnel are used in an affected area. Depending on the type of oil and energy of the impacted habitat, natural recovery may be most effective.

Mud flats should be monitored until traces of oil have disappeared, and injured biota have recovered. As direct estimates of recovery times for mud flat ecosystems are not available, it is presumed that three years would be necessary, as for subtidal soft bottom communities.

3.2.7 Estuarine and Marine Subtidal Habitats

Few case studies and no experimental studies are found in oil related literature on the injuries to or restoration of subtidal habitats following oil discharges.

Detailed studies of the shallow, subtidal habitats affected by the 1991 Gulf War oil discharges were conducted one year later. These studies were part of the 100 day cruise of the NOAA ship the *Mt. Mitchell*. Oil contamination of bottom sediments was visually observed by divers and samples were collected for chemical analysis. There was no evidence of large-scale sinking of oil in the nearshore subtidal habitats along the coastline of Saudi Arabia (Michel et al., 1993). Areas examined were those heavy hit by oil. If oil affected a habitat, it would be expected in these locations. In the 1983 *Norwuz* discharge in the Gulf, oil reportedly sank due to deposition of sediment onto oil slicks at sea by dust storms.

Estuarine and marine subtidal habitats are not often affected for long periods of time (long enough for restoration actions to be planned) by oil discharges unless oil adheres to particulate matter and sinks. This is most likely to occur in low salinity waters where water density is low (e.g., the *Tsesis* discharge in the brackish waters of the Baltic Sea, Sweden). However, restoration by capping or dredging to isolate or remove contaminated sediment can be employed much the same way as it is done to restore subtidal areas effected by chemical contamination.

Capping of the oiled subtidal habitat can be done by placing 0.5-1.0 m of clean sediment on top of contaminated sediment. The depth is dependent on sediment type (fine sediments contain the contamination more successfully) and the environment of the area. Sediment can be obtained from dredging projects or purchased from construction firms. Capping involves covering contaminated sediments to prevent their contact with surrounding water. The process is used when sediment removal is not possible. Contaminated materials are left in place and covered with enough material to prevent contaminated sediment - water interaction. Cap thicknesses in current practice in the United States for such purposes vary from 0.5 - 4.0 m thick (Truitt et al., 1989). An analysis by Thibodeaux et al. (1990) supports 0.5 m as being sufficient for undisturbed sediments but suggested that a thicker cap might be needed where animals excavate to greater depths. Malek and Palermo

(1987) suggest a design criterion of a 1.0 m thick cap as sufficient to prevent bioturbation-caused flux of contaminant.

Restoration techniques for contaminated sediments might involve sediment removal and subsequent treatment and disposal. During removal it is important to minimize the threat of additional environmental impact through resuspension of contaminants. It may also be important to temporarily divert water flow from the affected area while sediment removal is completed.

Except in the case of heavy, sticky oil adhering to subtidal habitats, natural recovery is recommended. When oil must be removed it can be dredged and replaced with similar clean sediment. When dredging is not feasible, the area can be capped to contain the oil contamination and prevent further mixing with the water column and/or effect on marine life.

Marine and estuarine benthic organisms will recolonize a capped or dredged area within one year following operations. Recovery would be expected in three to five years (Peterson, 1982; Yount, 1990).

It should be noted that, while use of bioremediation agents in open water has been attempted (*Mega Borg* and *Apex* barges in Texas), no detectable benefits could be demonstrated. Given high dispersion in open water, addition of bioremediation agents is not likely to be effective as a response and certainly not as a restoration option.

3.2.8 Riverine and Lacustrine (Freshwater) Shorelines

3.2.8.1 Case Studies of Oiling of Freshwater Shorelines

Riverine and lacustrine shorelines include freshwater rocky, cobble-gravel, sand and silt-mud shores. The terrestrial habitats bordering these shorelines often are vegetated with a variety of herbaceous plants, shrubs, and trees. Oil discharges in these freshwater systems, especially rivers, tend to have less of an impact than seen in marine and estuarine areas because the lack of tides minimizes the possibility of rafting up and beaching on shore and currents in river systems tend to carry oil downstream, limiting oil exposure to a single incident.

Few incidents of restoration in response to oil discharges in low energy river and stream habitats have been documented. Restoration following the NEPCO 140 oil discharge consisted primarily of removing oiled debris and vegetation. Bushes and shrubs were removed or cut back if oiled. These areas recovered more slowly than oiled areas that were left to recover naturally (Booth et al., 1991). Following the discharge in Little Panoche Creek, restoration actions included removal of contaminated soil and sediment. Sediment was either replaced with new material or cleaned and returned. A portable cleaning plant was employed following the *Amazzone* discharge (Huct et al., 1989). Sediment restoration appears to be effective in enhancing recovery based on this limited experience.

High energy rivers and streams are characterized by fast-flowing water, course-grained sediments consisting mainly of gravel and cobble, and little if any marsh habitat. Restoration activities for shorelines could include removal of oiled riparian vegetation and streambank soils. No case studies of restoration were found in the literature, possibly because of the expense of such action in a habitat most likely to recover naturally in a short period of time.

3.2.8.2 Experimental Studies on Freshwater Shorelines

Review of oil and non-oil restoration literature did not locate any experimental studies on riverine or lacustrine shorelines.

3.2.8.3 Freshwater Shoreline Restoration and Recovery: Summary and Conclusions

Oil discharge incidents that impact freshwater shorelines have been poorly documented. Restoration actions include natural recovery, removal, and replacement of sediment, cropping of oiled vegetation, flushing, sediment washing or incineration, agitation and bioremediation. For rocky and artificial shores, sand blasting or steam cleaning would be effective at removing oil, but should only be used when aesthetic or other non-biological values are more important than biological concerns (which are minimal on freshwater hard shores).

High energy, coarse-grained sediment shorelines of fast flowing river systems will recover within one week to one year (Booth et al., 1991) depending on the oil type and energy of the shorelines. Sediment removal along with cleaning or replacement has not been shown to increase recovery time. Low energy shorelines will require a longer period to naturally recover and sediment cleaning should be considered. Such decisions must also include consideration of use of the impacted area by the public, wildlife and birds.

Monitoring of shorelines impacted by oil should continue at least until the contaminant has been removed (naturally or mechanically). Areas where biological resources are significant (i.e., not including artificial shorelines where services are non-biological) should be monitored throughout the recovery period, approximately two to three years.

3.2.9 Riverine and Lacustrine (Freshwater) Unvegetated Bottom Habitats

3.2.9.1 Case Studies of Oil Discharges in Freshwater Unvegetated Bottom Habitats

As with riverine and lacustrine shorelines, the effects of oil on unvegetated bottom habitats and associated restoration actions are determined by the energy (i.e., flow or currents) of the impacted area.

Rivers and streams usually present conditions of high current flows and coarser sediments. These factors combine to give oil discharges a unique character, in which dilution and dispersion combine with relatively short-term oil persistence in bottom sediments. The literature indicates that oil can persist for weeks or months after a discharge depending on the oil characteristics, stream flow, and sediment characteristics (Vandermeulen, 1992).

Few case studies of oil discharges in flowing freshwater are available in the literature. Only one, an unleaded gasoline discharge into Wolf Lodge Creek in June 1983 involved a streambed. The others caused impacts to freshwater wetlands and are discussed in the review of freshwater wetlands above.

The Wolf Lodge Creek discharge resulted from a ruptured pipeline that released 25,000 gallons of unleaded gasoline. One month after the discharge, trapped gasoline in streambed gravel was released by raking the gravel with a bulldozer. Macroinvertebrate species were reduced for two weeks following streambed agitation. These same species reached advanced successional stages within six months. Surveys showed little difference in the recovery rates of raked and non-raked areas based on gross ecological measures. The agitation was considered beneficial, however, because it reduced possible sublethal chronic effects without causing substantial impacts (Booth, 1991).

Environments with relatively low water flow (lakes, ponds) are more likely to be impacted by oil discharges. Finer bottom sediments (silt, mud) correspond to a greater chance of the persistence of discharged oil. Effects may last for months or more and may involve the whole range of aquatic organisms (Vandermeulen, 1992). Lacustrine habitats may be restored by capping the impacted area or by removal of contaminated sediment followed by replacement of the substrate with new material or with the original sediment after cleaning. Capping can be completed by covering the contaminated area with up to 0.5 m of clean sediment to contain the pollutant and prevent its release to the water column.

Removal of contaminated sediment is most often completed by dredging. The most effective means is with an efficient hydraulic dredge that allows for removal of bottom sediment with the least additional impacts to the habitat (resuspension of oil, sediment, etc.). The benthic community is destroyed in such a process and has been shown to take two to three years to reestablish (Peterson, 1982). Many projects are conducted each year, primarily in the Great Lakes, by the US Army Corps of Engineers to dredge contaminated sediment and either treat and replace the sediment or dispose of it in confined disposal facilities. Recolonization usually occurs in one to three years (Yount, 1990).

3.2.9.2 Freshwater Unvegetated Bottom Restoration and Recovery: Summary and Conclusions

In high energy, course sediment (i.e., riverine) habitats, natural recovery is recommended unless oil persists. If natural recovery is inhibited or contamination is a concern for future injury, bottom sediments may be agitated to facilitate dispersion of oil from sediments. Unvegetated bottom sediments of lacustrine habitats (low energy environments) can be restored following oil discharges by allowing for natural recovery, capping, or removing and replacing the contaminated material. Again, the choice of actions should be made based on the need to remove or isolate contamination to prevent further injury. Monitoring should be conducted until biological species have recolonized, generally for three years.

3.2.10 Monitoring of Habitat Recovery

For every habitat, and for every restoration action chosen, some evaluation must be made of whether or not there is a return to conditions predating injury (i.e., whether it is successful), of the rate at which these processes occur and their extent, and of the stability and persistence of the recovered habitat. Each of these determinations requires a well-designed and executed monitoring plan, without which recovery cannot be properly established. Every habitat is a unique system, which will make it inappropriate to propose fixed monitoring plans for a generic habitat. Nevertheless, there are several general principles that apply to any such effort:

- Monitoring must occur over a sufficiently long period of time to document full recovery (or establishment of a new stable state) and to verify that the condition is stable;

- Monitoring should evaluate all components of the habitats. Floral and faunal coverage, biomass, composition, diversity, and physiology are all relevant parts that should be considered. Abiotic factors, such as soil qualities, should also be addressed. If continuing contamination is a problem, this too must be monitored;

- The progress of recovery should be compared with natural changes occurring in similar uninjured areas as control or reference sites;

- Sampling must be designed to provide statistically significant evaluation of changes in the recovering habitat and its components;

- The monitoring plan should be sufficiently flexible to permit mid-course alterations if the need arises; and

- Information must be reviewed, reported, and made available to scientists and managers.

It must be realized that in proposing a specific strategy for monitoring, the intent is to provide a basis from which to act and upon which an approximate cost may be estimated. Exact protocols to be used will be determined by experts in the field based on appropriate statistical principles, on the specific habitat affected, and, probably, on specific knowledge of local physical and biological conditions, that influence the time course of recovery. Also, there will be obvious local conditions that will alter the general plan. For example, many habitats undergo seasonal cycles that will make it meaningless to visit them at certain times of year.

The issue of what to measure will be habitat-specific and will be driven in part by the services performed by a given habitat. The functions of a saltmarsh, for instance, are quite diverse and monitoring of each function would in fact result in a very large program. Boland (1992) proposes that the monitoring program should seek to determine the abundance, biomass, age distribution, growth rates, and reproductive condition of all species influenced by the oil discharge. In practice, there will be a more limited program that measures these values for all key species, and perhaps for some additional species determined to be indicator species (sensitive species whose presence or absence indicates some stress) or target species known to play key roles in community structure (Boland, 1992). Remaining components of the community may be reduced to summary statistics such as diversity indices and total biomass and numbers, along with appropriate physical and chemical data.

While the goal of restoration is to return a habitat to the condition it would be but for the incident, that condition is difficult to determine after the fact. Therefore, control or reference areas must be selected that will establish what constitutes *recovery*. Since it is unlikely that any two sites will be exactly alike in all aspects, trustees must seek as control or reference areas sites that are comparable in such environmnetal variables as bottom or shoeline slope, water depth, tidal range, slainity, sediment composition, exposure to chronic pollutants. The closer the match between the affected area and the control or reference area, the more credible the results.

The probable time for habitat recovery is addressed under the separate discussions of each habitat. In most cases, this begs the questions of how recovery is defined. Most studies of habitat recovery fail to consider all of the components of a given habitat and many of them are not carried out to the point at which the habitat can reasonably be considered restored. If one takes too rigid a view, success is unlikely. A most reasonable view, stated by Ganning et al. (1984) is of "returning the ecosystem to within the limits of natural variability." This incorporates an important component of appropriate monitoring, which is to determine natural variability. It is not sufficient, however, to determine that a given habitat reaches a point at which it overlaps the distribution of unaffected habitats. Monitoring programs should be extended at least two years beyond this point of apparent recovery to verify that the condition is stable rather than transitory.

The natural variability of the impacted area will be an important determinant in the scope of the monitoring plan. Highly diverse and variable ecosystems will require large sample sizes to achieve a meaningful measure of the average condition. The general approach will involve some form of stratified random sampling (Boland, 1992; Stekoll et al., 1993). In most habitats there will be some basis for stratifying the area into components with differing characteristics, such as tide height. Separate random samplings are then taken within each of these subunits. In a single discrete area of restoration one might lay out several (e.g., five) transects within each stratum (or across all strata, if possible) and collect data from three quadrats randomly taken along each transect (within each stratum). It will be the responsibility of those designing the monitoring plan to verify that the numbers of samples collected are consistent with the level of variability in the habitat such that statistically valid comparisons may be made between impacted and reference sites and within the impacted site over time. At each quadrat, a determination should be made of percent cover by species and the numbers of each species present. Samples should be collected for determining biomass, growth rate, reproductive condition, or other variables appropriate to the habitat and season, as well as for determining physical variables. There are numerous other possible sampling plans. For example, Erwin (1988) proposes the use of line or strip transects for freshwater wetland monitoring.

Broader-scale phenomena will require a different approach. Wetlands, for instance, are generally considered important bird habitats. Evaluating habitat success will require observation over time. Crewz and Lewis (1991) suggest at least a 24-hour period of observation per monitoring visit. Similarly, evaluation of the importance of a seagrass bed as fish habitat will require a fish sampling program (e.g., Hoffman, 1991), as well as sampling programs for the epifauna and infauna.

The level of effort required to demonstrate recovery is difficult to quantify given the diversity of choices and habitats. Brooks and Hughes (1988) have proposed a standardized monitoring program for freshwater wetlands of 0.1-10 ha area (for larger areas they note the need for stratification). They suggest that a team of two professionals and two technicians could evaluate three such wetlands in a three day period (monitoring fewer at a time would be less efficient) and they propose that such monitoring occur six times per year. Data analysis and report preparation would involve added effort. While six times per year would be appropriate for the first year of a program, monitoring could probably be less frequent in ensuing years. Crewz and Lewis (1991) propose biannual sampling after a more intensely-sampled first year for evaluating saltmarsh restoration. For very slow recovery habitats such as coral reefs, annual visits may suffice after an initial establishment period of perhaps five years. It will be evident in the preceding years whether variation is large enough from one sampling period to the next to retain a more frequent periodicity of sampling.

3.3 Biological Natural Resource Restoration

Very little literature exist documenting restoration of shellfish, fish, or wildlife species following oil discharges. However, there is a vast non-oil related literature that is applicable to assessment of recovery and restoration of these natural resources, as reviewed below.

3.3.1 Shellfish

In general, various management approaches may prove useful for restoration of invertebrate populations. The few data available on restoration efforts involving invertebrate populations are described below.

3.3.1.1 Natural Recovery

Intertidal invertebrate communities appear to have long recovery times following disturbance. For example, natural recovery of a limpet, *Patella*, was observed following the *Torrey Canyon* oil discharge (Hawkins and Southward, 1992). Abundance and population structure were clearly abnormal for at least 10 years and recovery was estimated to take at least 15 years. This organism was particularly affected by dispersant spraying (i.e., complete mortality resulted). Snails, crabs, shrimps and echinoderms were very badly affected while survivors included hardier animals such as barnacles and topshells. Estimated recovery times for other species were not reported.

Estimated recovery times for marine soft-bottom benthos are on the order of 2-3 years (Peterson, 1978; Manci, 1989). Gore (1985) reported that benthic macroinvertebrates can recolonize a freshwater stream reach in a short period (75-150 days) but establishment of a stable community may take 300-500 days or longer.

3.3.1.2 Management Practices

Because of their reliance on nearshore habitats (i.e., estuaries, reefs, mangroves, etc.) invertebrates for which there are valuable fisheries like the Dungeness, blue, rock and Jonah crabs; Pacific shrimps, abalones, hard and softshell clams, bay scallops, oysters, periwinkles, blue mussels, and whelks are particularly susceptible to habitat loss, pollution, changes in freshwater flows, siltation, and other environmental problems. Overutilization has been at least partially responsible for depleting such species as Pacific razor clams, Pismo clams, abalones, oysters, and Pacific shrimp. Because many shellfish fisheries are close to large population areas, the likelihood of pollution problems is high. In addition to direct pollution impacts, excessive nutrient loads may increase toxic plankton blooms that cause red tides and paralytic shellfish poisoning. Atlantic coast and Gulf of Mexico oyster and hard clam harvests have been severely reduced by pollution, disease, salinity changes, and habitat losses. Louisiana alone loses an estimated 35,200 acres of coastal wetlands habitat each year. In addition, marine mammals also feed on some of these species. On the Pacific coast, sea otters have depleted abalone and sea urchin stocks, particularly in California (NOAA, 1991b).

As reported in NOAA (1991b) many methods are used to harvest the invertebrate species. Commercial and sport divers gather abalones, particularly in southern and central California. Fishermen in small boats dive, dredge, and tong for oysters and rake hard clams. Recreational clammers dig Pismo clams on sandy beaches in central California and razor clams in the Pacific Northwest. Trawlers and divers take sea urchins off the New England and northern Pacific coasts. Commercial and recreational crabbers fish with pots, traps, trotlines, dredges, and dip nets for blue, rock, and Jonah crabs on the Atlantic coast and for Dungeness crabs on the Pacific coast. Pacific shrimps are harvested with pots and trawls. Other species, such as blue mussels, are both cultured and harvested from the wild.

Because these species frequent nearshore waters, they are not included in federal fishery management plans. Some are managed under regional, state, and/or local authority. Typically, size limits are used to protect molluscan and crustacean resources from overutilization, whereas area closures, bag limits, and catch quotas are employed for other groups.

The state of Alaska Department of Fish and Game (ADFG, 1985) described some of the standard techniques used in managing their invertebrate fisheries as follows:

- Tanner Crab - impose season and gear restrictions, size and sex limits and specify harvest levels; minimize mortality on female crabs; and assure full female fertilization by providing adequate number of mature males for breeding;

- King Crab - size, sex, season, area, gear restrictions and a flexible quota system;

- Dungeness Crab - males only fishery; fishing season timed to protect crabs during molting and softshell periods; and gear restrictions; and

- Shrimp - gear restrictions; guideline harvest levels determined each season based on abundance indices from trawl surveys; no closed season for pots; trawl fishery regulated so that closures would correspond to egg-hatching periods in the spring months.

Steele and Perry (1990) described the standard management practices associated with blue crab fisheries as:

- Minimum size limits;
- Protection of female crabs - illegal to keep or sell adult female crabs with eggs; and
- Fishing methods and gear restrictions.

The goal of management is to maintain fishable stocks. The application of similar or stronger management practices could be used to enhance depleted stocks. Steele and Perry (1990) additionally noted that variations in salinity, temperature, pollutants, predation, disease, habitat loss, and food availability are the major factors affecting blue crab survival. Thus, elimination of pollution and habitat loss could also result in enhancement and/or restoration of blue crab populations.

One of the most serious instances of chemical pollution affecting a blue crab fishery occurred in Virginia and was associated with the release of the chlorinated hydrocarbon kepone into the James River from the 1950s to late 1975. The annual mortality of young and adult blue crabs due to kepone remains unknown. However both commercial landings and juvenile abundance were lower in the James River than in the York or Rappahannock rivers for a 15-year record. The ban on use of a similar chlorinated hydrocarbon, DDT, may have resulted in the recovery of the blue crab resource in the late 1970s.

Van Engel (1987) noted that the blue crab is characterized by the annual production of a large number of young, interannual fluctuations in production, rapid growth, early attainment of maturity, high mortality and short life span. Because of these characteristics, the blue crab should have both a quick recovery if overfished and good natural recovery after manmade or natural disasters.

Maigret (1985) reported that populations of two species of rock lobster (nearshore and deepwater) were restored to formerly abundant levels following cessation of fishing. Both stocks were at very low levels between 1970-75 due to overexploitation. After 1975, political events closed the fishery and the populations recovered and stabilized. Temporary closure of a fishery may thus be sufficient to restore lobster populations under certain conditions.

3.3.1.3 Culturing

Shellfish, in general, are actively cultured and seeded to enhance the wild stock (Petrovits, 1985). Quahog fisheries are usually enhanced in two ways by relaying (i.e., transplanting) clams from underutilized or crowded flats and by culturing hatchery-reared seed until they reach 25 mm at which point they are broadcast to the fishery. Quahog populations are subject to negative effects from overcrowding such as slow growth and high mortality before reaching legal size. Thinning and transplanting quahogs to less populated areas should, therefore, maximize growth and reduce mortality and have a positive effect on population size.

Kelley et al. (1984) described techniques for collecting bay scallop spat from the field by using old onion bags filled with fine mesh netting. After reaching 10-20 mm the scallops were transferred to floating cages where they were grown until they reached 40-50 mm. Afterward the scallops were scattered in good growout areas. This or similar culturing techniques have been adopted as enhancement tools by several states. Regrettably, the utility of the technique in actually enhancing bay scallop populations has not yet been conclusively demonstrated. Walsh (1984) concluded that current scallop aquacultural techniques hold little promise to enhance or support recreational or commercial bay scallop fisheries.

Gaines and Ross (1984) summarized actions needed to improve the bay scallop fishery in Massachusetts as:

- Increased research on larval behavior, adult ecology, and life history study;
- More regulations with better enforcement; closed seasons for draggers; establishment of controlled areas; limited entry and limited effort (five day week);
- Environmental enhancement such as thinning beds, returning shells to the water, predator control, protecting breeding populations, and control of *Codium*;
- Life history and culture-based remedies such as using hatchery seed to supplement natural set, setting out spawning stock, development of nurseries, moving seed offshore, undertaking mariculture at the local level, and spat collection; and
- Education in the form of public information and open communication among scientists, fishermen, and officials.

Several attempts were made to enhance red abalone, a commercially important species in southern California. Hatchery raised juveniles were released in several kelp forests. However, the result was a low rate of survival (Tegner and Butler, 1985; 1989). There is evidence that hatchery raised abalone are more vulnerable to predators (Schiel and Welden, 1987). Other attempts at restoration or enhancement have taken advantage of the short larval life and consequently short dispersal distance of planktonic abalone. Mature red abalone were transplanted to an area with a natural gyre where larvae may be entrained (Tegner and Butler, 1985). A relatively large recruitment occurred during the year following the transplant.

Culturing and stocking of larval and juvenile target species should continue to be pursued to restore invertebrate populations to coral reef habitats. There have been advances in mariculture research on the culturing and growth of giant clams (Price, 1988) and Caribbean queen conch (Berg, 1976), and success is being reported for spiny lobsters (Prescott, 1988), green snail (Yamaguchi, 1988), top shell (Gillett, 1988), and the black lopped pearl oyster (Sims, 1988).

Munden (1974) described the North Carolina Pamlico Sound oyster restoration project that was designed to restore the natural oyster producing grounds impacted or destroyed by Hurricane Ginger in September 1971. The objectives of the project were to reseed areas with shell stock and/or marl to compensate for the mortality of small oysters and to reestablish base rocks to prevent loss of the traditional producing grounds. All plantings were in areas protected by shoals, coves, leeward shorelines and/or bays to reduce losses caused by winds, and were in areas with a history of high production of good quality oysters. Three months after planting, samples of planted materials were collected to determine spat set. Seed oysters were subsequently planted in areas with low spat set. The results of the restoration effort were not described. However, prior to the hurricane, oyster production in the Sound had been increasing, perhaps, in part, to restoration efforts begun in 1970.

3.3.1.4 Stocking

Brinck (1988) reported on efforts to restore crayfish populations in Sweden after the introduction of a crayfish plague caused heavy mortality. To slow the spread of disease and protect the crayfish fishery, the Swedish authorities took the following steps:

- Prohibition of live crayfish importation;
- Prohibition of removal of live crayfish from infested waters;
- Fishing gear, boats, boots and other equipment were disinfected;
- Restriction of fishing seasons and introduction of strict minimum size;

- Initiation of a program investigating the possibilities of finding a resistant substitute for the native crayfish species, which later expanded to studies of the resistant North American signal crayfish;

- Establishment of a research program aimed at increasing knowledge of the causative agent and its Swedish host;

- Implantation of U.S. adult signal crayfish into a selection of Swedish lakes;

- Stocking of imported U.S. adult signal crayfish replaced by stocking of juveniles cultured in Sweden; and

- Introduction of signal crayfish was restricted to those regions where the plague was well established.

A period of sixty years elapsed between banning the import of live crayfish and the decision to replace the native species in plague areas with the signal crayfish. Over that period, Sweden experienced a 90% reduction in production of native crayfish. Within the last ten years, a flourishing production of signal crayfish was established. Under economic pressure from fishermen, the Swedish authorities chose not to wait for development of a resistant wild strain when an alternative solution (i.e., introduction of a non-native species) appeared possible.

In recent years, the plague has spread to the Turkish fishery. Based on the experience gained in Sweden, two actions for preserving the fishery were recognized live with the plague and wait for resistance to appear in wild species, or introduce a species of crayfish that is resistant to the plague and is capable of adapting to the new environment. The situation in Turkey may be different since the causative organism loses its viability in soda lakes and several large Turkish lakes will thus retain their native crayfish populations. There is a growing resistance among scientific experts to introduce non-native species to an ecosystem. Although a final decision has not yet been reached, consideration was focusing on attempt to find resistant Turkish crayfish and setting up a breeding project for such individuals.

3.3.1.5 Rehabilitation by Oil Removal

Burger and Gochfeld (1992) described the results of gently washing fiddler crabs after they were exposed to oil during a 1990 discharge in the Arthur Kill (between New York an New Jersey). Changes in behavior were reported over a ten-day period for oiled crabs that emerged prematurely from their burrows. Behavioral changes were compared between crabs that were washed with freshwater and those not washed. Locomotion, aggression, balance, and burrowing behavior were examined. Unwashed crabs improved significantly on only one of twelve behavioral tests, while washed crabs significantly improved in four tests relating to movement, defensive behavior, and burrowing. The washed crabs exhibited a greater improvement on ten of twelve tests when compared to unwashed crabs. Washed crabs showed greater improvement in their ability to find and construct their own burrows. These experiments indicate that oil removal improves the behavioral performance of crabs and suggests that, under some circumstances, the immediate flushing of salt marsh creeks by uncontaminated tidal water may decrease behavioral effects on crabs. Since burrowing behavior is important for survival, any improvement in this behavior would improve recovery of the crabs.

3.3.1.6 Enhancement through Reconstructed Wetlands

The few data available on invertebrates of constructed marshes have demonstrated considerably lower abundances or vastly different species than present in reference wetlands. Rutherford (1989) found similar epibenthic species in a 4-year old constructed marsh but greatly reduced densities. Cammen (1976a, b) reported significantly different infaunal species in constructed wetlands along the coast of North Carolina. Sacco et al. (1987) noted that after 15 years the same North Carolina site showed a 10-fold increase in densities and high similarity with the infauna of natural marshes. Species composition does not always become similar over time (Moy, 1989; Saccor, 1994). Sacco (1994) further noted that six constructed marshes had similar fauna 1-17 years after construction but uniformly lower abundance.

A study in Texas found consistently lower densities of brown shrimp and grass shrimp but equal densities of blue crab in planted marshes after 5 to 6 years relative to natural reference marshes (Minello et al., 1986). West (1990) noted difference in invertebrate community structure among creeks in natural and created brackish water marshes.

3.3.1.7 Harvest Refuges

A comparison of areas that are protected from exploitation either by regulation or inaccessibility show that resident species, such as lobster and abalone, are more abundant and reach a larger size in protected areas (Cowen, 1983; Cole et al., 1990). As is seen for fish populations, establishing harvest reserves is a promising technique for enhancement of invertebrate populations impacted by overexploitation or environmental degradation.

3.3.2 Fish

Restoration of fish populations is accomplished when baseline populations are present and productive and normal age distributions are achieved (Koenings et al., 1989). Efforts to restore fish populations are dependent on identifying sources of reduced survival, and continued monitoring to assess their disappearance or persistence. Relatively little work has been completed in restoring fish populations following oil discharges, although ongoing studies following the *Exxon Valdez* oil discharge will provide valuable guidance to future restoration work.

Some of the potential effects of oil on the fish or fishery include:

- Depressed feeding (Williams and Kiceniuk, 1987);
- Decreased swimming activity and increased mortality (Berge et al., 1983);
- Mortality to eggs and larvae (After the *Argo Merchant* discharge 20% of the cod eggs and 46% of the pollock eggs in the discharge zone were dead. During the *Torrey Canyon* discharge 90% of the pilchard eggs in the discharge area were killed. However, compared to the naturally high mortality rates of fish eggs these losses would be hard to detect in the commercial harvest. Following the *Amoco Cadiz* discharge, a one year old class of flatfish was thought to have been reduced.);
- Exclusion of fishermen from the fishing grounds and other disruption of fishing that can change the population balance to date (e.g., salmon overescapement in Prince William Sound after the *Exxon Valdez*);
- Fouling of fishing gear;
- Tainting of fish (i.e., change in flavor or smell) and the public's fear of tainting;

- Mortality or other effects of non-motile inshore species, such as rockfish (e.g., EVOS Trustees, 1992a);

- Mortality or other effects of fish maintained in mariculture enclosures (where escape of fish is prevented) (e.g., the *Braer* oil discharge off the Shetlands affected salmon in mariculture enclosures); and

- Sublethal effects such as:

  - Fin erosion;
  - Ulceration of the integument;
  - Liver damage;
  - Lesions in the olfactory tissue;
  - Reduced hatching success;
  - Reduced growth;
  - Change in egg buoyancy;
  - Malformations that interfere with feeding;
  - Arrest of cell division; and
  - Genetic damage.

In the absence of a sufficient published literature for oil-related restoration methods for fish populations, a summary of proven and unproven methods for restoring fish populations is presented below. Such methods are generally applicable to restoration of oil-impacted fish populations.

3.3.2.1 Natural Recovery

Natural recovery is effective for some natural resources. In the case where a fishery is allowed to recover from a fish-kill by natural replacement without the help of restocking, the major problems are:

- Loss in commercial fish revenues associated with a reduction in catch;

- Drop in market value due to a perceived injury (e.g., tainting) by the consumer;

- Loss in recreational opportunity; and

- Lost passive use value (i.e., value of a fishery independent of use) resulting from a fish kill.

Baker et al. (1990) reviewed the natural recovery of cold water marine environments following an oil discharge. They noted that human uses of a discharge-affected area generally resume as soon as the bulk of the oil is removed. Human uses include both commercial and recreational fishing. Although these activities may resume rather quickly, the availability of human services is not necessarily related to biological recovery, which progresses more slowly.

Commercial and sport fishermen are generally excluded from fishing grounds where oil is floating on the water because of the risk of fouling fishing gear and of tainting. Often it is possible to fish in areas unaffected by oil and commercial fishing can continue even after a major oil discharge. This was the case for larger fishing vessels in Brittany, France, following the wreck of the *Amoco Cadiz* (Fairhall and Jordan, 1980).

Fairhall and Jordan (1980) feel that fish stocks are rarely directly affected by oil discharges and a fishery in an area that has been exposed to oil can be reopened as soon as the area is free of oil. Recovery of use of the area usually takes place in a matter of days or weeks and is independent of the biological recovery of the injured ecosystem. On the other hand, when the fishery resource itself is injured, fishery losses will persist until exploitable stocks are restored. Compensating for losses of catch may require deliberate restocking and a delay of 2 to 10 years, depending upon the age at which new stocks reach exploitable size.

Animal communities from cold water ecosystems tend to be less stable than those from lower latitudes owing to the harsher environmental conditions. As a consequence, there can be considerable natural variability in community species composition from year to year. Animals from polar and subpolar regions tend to adopt reproductive strategies that involve either viviparous (live-bearing) or oviparous (direct development from egg to an apparent miniature adult) development. Since such strategies are associated with greater parental care, with fewer offspring per reproductive cycle, these populations are less likely to recover from major environmental injury as rapidly as those more southerly species producing vast numbers of planktotrophic larvae. Although Baker et al. (1990) do not provide estimates for time of natural recovery of fish populations, they present recovery times for various environments. They state that past experience has shown that exposed, rocky shores in the north usually recover in two to three years. Other shorelines show substantial recovery in one to five years with the exception of sheltered, highly productive shores (e.g., saltmarshes) that may take 10 years or more to recover. Subtidal sand and mud systems usually recover in one to five years, but they can take as long as 10 years in exceptional cases. The authors also conclude that there is no evidence that sublethal effects are of any longer-term ecological significance.

Baker et al. (1990) note that kills of adult fish from exposure to oil are rare. Mobile fish species appear to be able to avoid oiled areas following a discharge. However, non-mobile, inshore species may be killed or otherwise affected and fish in mariculture enclosures cannot escape and are likely to be killed. For offshore species, there could be a heavy loss of pelagic eggs and fish larvae if present at the time of a discharge. In most cases, this mortality has had no detectable impact on the fish stocks available to the fishing industry. Annual recruitment to these stocks fluctuates naturally and the size of catchable stock is determined primarily by the activities of the fishing industry (i.e., overfishing) and climatic changes. The article fails to mention that anadromous species, such as salmon stocks, or shoreline spawners, such as Pacific herring, can be adversely affected by oil discharges that occur in the near coastal and coastal habitat, particularly during the spawning season. The *Exxon Valdez* oil discharge may have affected wild pink and chum salmon, as well as spawning herring, in Prince William Sound (Exxon Valdez Oil Spill Trustees, 1992a). Various amounts of oil were deposited in the intertidal areas where up to 75% of spawning occurs. Salmon eggs deposited in 1989 and in subsequent years have been contaminated and egg mortality documented. A higher occurrence of somatic, cellular, and genetic abnormalities have been noted among alevins and fry in oiled areas. However, population impacts are still unknown. Wild salmon fry consumed oil contaminated prey, which caused reduced growth and lower fry-to-adult survival. Predators targeted these smaller, slower growing fish. Reduced growth and survival during the early marine period may have caused the decline in returning salmon numbers in 1990 (15 to 25 million fewer fish). There is speculation that recently-detected genetic injuries may further reduce the productivity and fitness of wild salmon in Prince William Sound for many years (Exxon Valdez Oil Spill Trustees, 1992a).

Following the wreck of the *Amoco Cadiz*, there was an immediate kill of several tons of rockfish at the site. Generally, however, fish appear able to leave an oiled area. During the period when oil slicks were in the Santa Barbara Channel following the 1969 blowout, fish shoals were observed from the air by professional spotters in areas not covered by oil and no heavy mortality of fish was recorded (Abbot and Straughan, 1969). After the *Tsesis* oil discharge in the Baltic and the wreck of the *Betelgeuse* in Bantry Bay, Ireland, herring (and their sprat) migrated through the oiled areas and spawned normally (Linden et al., 1979; Grainger et al., 1980).

The loss of fish eggs and larvae from oil exposure must be weighed against the normal mortality. Only a small number of larval fish survive to an age when they reach an exploitable size. Additionally, most fisheries are based on fish of various ages and if the size of one year-class is reduced, that is unlikely to have more than a marginal effect on the commercial catch. Nonetheless, the wreck of the *Amoco Cadiz* may have resulted in a significant reduction in a one year old class of flatfish.

Calculations made by Johnston (1977) suggest that, even a catastrophic oil discharge (i.e., 400,000 tons) in the North Sea would be responsible for a loss of only 13,000 tons of fish. Since the annual commercial catch is 4.36 million tons, this shortfall would be hard to detect, particularly against the natural fluctuations in fish abundance.

Gundermann and Popper (1975) described some aspects of the recolonization of coral rocks by fish in the Gulf of Aquaba following a chemical discharge. As a result of an accident, a limited strip of the coast of Eilat was affected by pesticides and chemicals that killed all fishes. The morning after the discharge, masses of dead fish were observed floating on the water surface or lying on the sea bed. No live fish were seen in the poisoned area down to a depth of 15 m, and few survivors were found in the bordering zone at a depth of 15 to 25 m. Prior to the discharge, the area supported an abundant and diverse fish fauna. The area was observed monthly for the following year to study the recovery of fish populations. The study also included observations on growth rate of fish and population size. Recovery of fish populations was complete 10 to 12 months after the discharge, primarily by the recruitment of juveniles. The community appeared very similar to what it had been prior to the discharge both in number and composition of fish species. The rapid recolonization may be a result of the relatively small size of the contaminated area and the survival of most invertebrates that constitute an important part of the biotype of the fish. It is likely that under less favorable conditions it would take a longer time for a fully destroyed population to recover. During 1971 and 1972, the study area in the Gulf of Aquaba was exposed to pollution by many oil discharges and other materials from the nearby harbor and a new oil jetty. When visited in 1972, the area was devastated beyond recognition. Most of the corals were dead and no coral fish were found. Recovery would require reestablishment of the coral reef before fish populations could recover.

Jernelöv and Linden (1983) noted that the discharge of crude oil from the tanker *St. Peter* in 1976 affected a coastal area in Columbia and Ecuador, contaminating large parts of several mangrove swamps. Acute effects included observation of dead and decaying fishes at several sites two months after the discharge. During the following year, adult fish had returned to most sites suggesting that a mechanism for natural recovery was migration from unaffected parts of the mangrove swamp as soon as oil toxicity had disappeared. The oil discharge apparently affected the yearly migration route of skipjack and yellowfin tuna. They bypassed the affected site and were found farther north suggesting that migratory tuna avoid oil contaminated areas (or areas depleted of food because of oil impacts).

Although unrelated to a discharge, Brock et al. (1979) provided insight on the time required for unassisted recolonization of a small coral reef patch in Hawaii. A small, isolated reef was poisoned with rotenone to remove all fish occupants and the natural rate of recovery was subsequently studied. Following a 1977 fish collection, recolonization was studied for one year. Recolonization proceeded rapidly and occurred primarily by juvenile fish well beyond larval metamorphosis. Within six months of the collection, the trophic structure was reestablished. The MacArthur-Wilson model of insular colonization described the recolonization process and predicted an equilibrium situation in less than two years. The recolonization data suggested that, chance factors may explain the colonization process on a small scale, but a relatively deterministic pattern emerged when considering the entire reef. Thus, the authors concluded that at the community level, the fish are a persistent and predictable entity. It should be noted that the impacted area was very small, on the scale of meters. Fast recovery is unlikely to be possible for large scale impacts.

3.3.2.2 Modification of Management Practices

3.3.2.2.1 Traditional Methods

Historically, the most widely used and viable technique for enhancing freshwater and marine fisheries is to use a spectrum of regulations to control harvest. However, most authors have concluded that these restrictions are limited in power to increase the resources available for harvest, or to affect the temporal or spacial distribution of these resources (Buckley, 1989). The standard techniques to enhance fisheries include:

- Size limit (i.e., limit age of fish taken);
- Catch quotas (i.e., limit size of fishery);
- Seasonal fishing restrictions;
- Selectively reduce harvest of injured stocks;
- Limit area fished; and
- Restrict gear efficiency.

The Green Bay Rehabilitation Story (Smith et al., 1988) is an example of the successful use of catch quotas to increase the abundance of an important commercial fish. Lake Michigan's Green Bay has a long history of misuse and overexploitation (Smith et al., 1988). The bay's problems started in the 1800's when commercial fishermen netted abundant stocks of fish; lumberjacks cleared the region's mature forests; and cities, industry, and agriculture grew to dominate the watershed. These activities resulted in degraded water quality, destroyed fish and wildlife habitat, and reduced fish populations. The fisheries were further destabilized by the introduction of exotic species such as smelt, lamprey, carp, alewives, and pink salmon.

Problems continued unabated until the late 1960s and early 1970s. In the early 1970s, PCBs were discovered in the water, sediments, and fish of Green Bay. Since 1970, $338 million was invested in wastewater treatment facilities resulting in decreased biological oxygen-demanding water and suspended solids. Mean summer concentrations of phosphorus also decreased and the abundance and composition of the benthos improved. The Wisconsin Department of Natural Resources established annual quotas for commercial yellow perch catch. With the improvement in water quality and concurrent fish management actions, the fishery has made an astounding recovery. The annual quota for commercial catch was raised from 200,000 pounds in 1983 to 400,000 pounds in 1987. The annual sport harvest is estimated to be more than 250,000 pounds. A rapidly growing walleye fishery, initiated by a mass stocking program, developed in the adjacent Fox River. Furthermore, the levels of PCBs in Lake Michigan fish are declining.

Although the above article (Smith et al., 1988) reports success for the restoration of yellow perch, no mention is made of the once abundant whitefish, herring, sturgeon, lake trout, chubs, suckers, and catfish that also declined because of pollution. Restoration or rehabilitation of these traditional Green Bay fisheries remains a goal for the future. The report does demonstrate that catch quotas are a successful rehabilitation technique for increasing abundance of certain fish species. In particular, yellow perch responded to decreased commercial catch combined with improved water quality by dramatically rebounding in abundance. It is important to note that this species, although reduced to low levels, was not eliminated from the natural environment.

3.3.2.2.2 Harvest Refugia

As noted by Davis and Grant (1989) traditional management controls on marine fisheries are exercised through limits on individual fish sizes, seasons of harvest, catch limits, and gear restrictions to protect reproductive stocks. Few nearshore fisheries are able to sustain high yields using traditional species-specific management strategies. Davis (1989) feels that designated harvest refugia or fishery reserves should be evaluated as management tools to restore, enhance, or sustain fisheries.

The effectiveness of multispecies harvest refugia in marine fisheries is not yet well tested. However, evidence for coral reefs in the Philippines (Alcala, 1981; 1988) and from a temperate ecosystem in New Zealand (Jeff, 1988) provides encouragement that such refugia may be extremely effective fishery enhancement tools.

In the Philippines, eight small areas (8-10,000 ha) were excluded from harvest for 3-10 years. The area with the longest period of protective management, a 750-m long segment of reef, was closed to fishing in 1974. Mean harvest rate was 0.8 kg man-day^{-1} before closure. Within 2 years, the mean harvest rate from areas adjacent to the closed zone had tripled and over a 5-yr period, the sustained yield of fish per area from adjacent zones was one of the highest reported for any coral reef (16.5-24 MT km^{-2} yr^{-1}, Alcala, 1981; Russ, 1987). After 10 years, the reserve boundaries were violated by fishermen and two years later yields in the entire area had declined more than 50%.

Harvest was prohibited in the 547-ha Leigh Marine Reserve in New Zealand for 11 years. Fish populations within the reserve have increased 2.5 to 20 times the densities in similar adjacent habitat, and both commercial and recreational fishermen believe that the reserve has increased their catches in adjacent areas. Regrettably, conclusive fishery yield data from areas adjacent to the reserve are currently unavailable.

Single-species sanctuaries for spiny lobster have proven to be an effective management tool. In both Florida and New Zealand, closing moderately large areas (100 to 1000 km^2) of juvenile lobster habitat to harvest increased adjacent adult populations and overall yields to the fisheries (Booth, 1979; Davis and Dodrill, 1980). A 190-km^2 marine park at Dry Tortugas, Florida, also serves as an adult lobster harvest refugium. It provides larval and juvenile recruits to adjacent and distant zones, protects genetic diversity of stocks, and serves as a site for research on natural mortality rates and environmental carrying capacity (Davis, 1977).

Selection of refuge sites (i.e., size and location) should be based on protecting ecologically discrete zones that are naturally buffered from environmental perturbations and that can produce larval and juvenile recruits for harvest in adjacent zones. When attempting to restore fish populations in discharge-affected areas, refuge locations may prove essential and should take advantage of natural processes that will promote dispersal and recruitment to the affected area. Optimum refuge design will most likely require compromises among the ecological requirements of several species. Empirical evidence should consequently be gathered to ensure that the most critical natural resources are not threatened by such compromises. Selection of the harvest zones and adjacent refugia must also involve the fishing community. In particular, the boundaries between zones must be recognizable and enforceable. Thus, law enforcement staffs and patrol activities are a necessary long-term expense when establishing refugia.

3.3.2.3 Stocking

Stocking may be used as a restoration, enhancement, or rehabilitation tool for anadromous and/or freshwater species. The successful enhancement of anadromous fish is frequently linked to artificial production in hatcheries. This technique supplements natural levels of recruitment of juveniles, which can dramatically increase the natural resources available for harvest if the other factors necessary for survival are not limiting in the natural environment. Stocking is a viable restoration technique provided a genetically suitable stock can be obtained. Stocking can occur at different life stages. The exact stocking strategy adopted will depend on the short- and long-term goals of a specific project.

McNeil et al. (1991) cautions that the continued production of hatchery fish for enhancement of salmon fisheries is currently being challenged by environmentalists. The major problem is the over-exploitation of the naturally-reproducing populations in mixed stock fisheries. Also, there is a concern that the stocked fish, which are released at a large size, may out-compete natural stocks for food. Opponents to continued operation of hatcheries are urging that priority be given to conservation of remaining wild genomes. A scientific assessment of naturally spawning salmon populations is now being conducted in the Columbia basin and other watersheds. Populations of a given species homing to individual sub-basins potentially qualify for listing as threatened or endangered under the Endangered Species Act.

Recent concern over maintenance of genetic variability portends a changing role for salmon hatcheries. Hatcheries will continue to operate but will foster reduced harvest of hatchery fish in mixed stock fisheries and greater harvest in terminal fisheries. Thus, surplus hatchery fish are harvested after wild and hatchery fish become segregated. In cases where natural stocks are reduced, the goal of hatcheries is likely to shift from enhancement to genetic conservation. Attention will be given to maintenance of genetic integrity of genomes of wild populations and avoidance of introgression of genes from hatchery populations produced for fisheries enhancement. This is a definite shift in public policy away from enhancing fisheries and towards conservation of indigenous genotypes. As noted below, when attempting to restore fish populations destroyed by oil discharges, particular attention must be focused on the importance of genetically distinct stocks.

Harrel et al. (1990) describe the next twenty-five years of striped bass and striped bass hybrid culture. During the past, substantial increases in recreational and commercial fishing, habitat alterations, and water pollution have reduced striped bass populations in inland and coastal waters. These populations have historically been managed by regulations that control seasons and restrict harvests, including moratoria on harvesting. Large minimum size, restricted seasons, and reduced creel are the rule along the coast and inland.

Declines in stock have resulted in a rapidly growing aquaculture and stocking industry. Future research will focus on nutrition, domestication of brood stock, evaluation of inland and private fisheries, strain selection, genetic manipulation through selective breeding, hybridization, polyploid induction, possible recombinant genetics, and production enhancement through reproductive and growth physiology. The authors anticipate that striped bass hatcheries cannot be required to maintain fishable levels. Hatcheries are expensive to build and maintain and the practice can alter the gene pool. However they provide a powerful tool for restoration of viable populations in inland reservoirs and in estuaries.

3.3.2.3.1 Importance of Genetically Distinct Stocks for Restoration

Stocking of hatchery-propagated domestic and/or wild strains of fish, or both, is an important tool for the restoration of fish populations. Krueger et al. (1981) described two possible stocking strategies, stocking with as many wild stocks and interstock hybrids as possible to maximize the genetic variability introduced into new environments or introduction of only those stocks whose native environments closely match the body of water to be stocked. Krueger et al. (1981) identified genetic monitoring as an important element of stocking programs. Evaluation of stocks is important because agency efforts expended in stocking strains with low survival or poor reproduction are largely wasted if the goal is reestablishment of a fish population.

3.3.2.3.2 Stocking Strategies

The following are strategies which may be used for stocking fish:

- Stock with eggs;

- Stream side incubation boxes followed by stocking with parr, fry, or smolts;

- Remote egg-takes and incubation at existing hatcheries followed by stocking with parr, fry, or smolts;

- Fry rearing (Fry plants are a proven method used by FRED Division of ADFG to rehabilitate and enhance sockeye salmon stocks); and

- Stock with yearlings.

The following review describes some of the long term projects for anadromous and freshwater fish species that use stocking as a key element for restoration.

Restoration of Striped Bass to the Kennebec River, Maine

Prior to the 1920's and 1930's, native spawning populations of striped bass were known to occur in the Kennebec/Androscoggin River estuary, Maine (Squires and Flagg, 1991). The native spawning population was believed exterminated due to heavy industrial and municipal pollution. The pollution resulted in dissolved oxygen levels that routinely dropped to zero throughout late summer and low river flow periods. Extensive pollution abatement efforts of the early 1970's brought about dramatic improvement in water quality. Squires and Flagg (1991) reported the following necessary criteria to support a native stock of striped bass:

- A minimum of 12-15 miles of unobstructed river flow;

- An average depth of 15 feet; and

- A minimum dissolved oxygen level of 5 ppm.

Maintenance of good dissolved oxygen levels from 1977-1981 prompted the Maine Department of Marine Resources to initiate an experimental striped bass restoration program (Squires and Flagg, 1991). In 1982-1983, wild young-of-the-year (YOY) striped bass were captured from the Hudson River and transferred to the Kennebec River. Because only small numbers could be obtained from seining wild fish, the program was shifted to hatchery production in 1984. From 1985-1990 hatchery produced fry were raised to fall fingerling size and stocked into the Kennebec River estuary. From 1982-1990 a total of 252,793 fall fingerlings were stocked, ranging from a low of 319 in 1982 to a high of 66,000 in 1988. In 1987, 26 wild YOY striped bass were collected at three locations, representing the first documented spawning success of striped bass in the Kennebec/Androscoggin River estuary in over 50 years. Wild YOY striped bass have been collected each consecutive year from 1987-1990. In addition, ichthyoplankton surveys on the river since 1988 have yielded low numbers of striped bass larvae. Squires and Flagg (1991) conclude that stocking hatchery-reared striped bass juveniles can be used to reestablish spawning stocks in reclaimed historical spawning habitat. Although the full restoration of the striped bass population has not yet occurred, the modest returns to date are encouraging and should be viewed as a positive contribution to the resource.

Many fish restoration projects have involved a combination of several of techniques. Marsden (1987) described the restoration of native lake trout to Lake Ontario through a long range plan that combined stocking (yearlings) with water quality improvement efforts, a cessation of commercial fishing and an intensive program of lamprey control. Lake trout, an important component of the multimillion dollar Lake Ontario recreational fishery, were originally eliminated from their natural habitat in Lake Ontario through a combination of overfishing, lamprey predation and habitat degradation (i.e., organic debris and siltation). In particular, the addition of silt to the spawning beds initially reduced the ability of the trout to recover from overfishing and lamprey control. Stocked yearling trout from a variety of genetic strains were able to establish a sufficiently abundant population to support a significant sport fishery. The ultimate goal of the restoration effort, however, was to restore a naturally spawning population to the lake. A 1986 intensive sampling program indicated that hatchery-reared lake trout stocked into the lake could survive, mature, and produce offspring. In the future, additional spawning shoals will be checked for fry to assess the health of various spawning grounds, determine their parental stock, and improve spawning habitat where necessary.

Spurrier and Morse (1988) noted that lake trout in Lake Superior were similarly devastated by heavy exploitation and the invading sea lamprey. The population is currently being rehabilitated (restoration is not yet a reasonable goal) through concurrent efforts of lamprey control, stocking of yearling lake trout, and limiting commercial harvest to fish taken under special permits. Lamprey control has held the lamprey to 10% of its peak abundance in Lake Superior since the early 1960's. Lampreys are still a major lake trout mortality factor. The population is responding with improvements in number, size, and age structure. Stocked fish make up the bulk of the population although evidence of natural reproduction was documented recently.

In addition to lake trout, the naturalized rainbow trout population is supplemented by stocking and chinook salmon are stocked in the lake. Coho salmon were stocked until 1974 and still appear in Minnesota waters. The suspected source of these coho is stocked or naturally-produced fish from the south shore of Lake Superior.

Spurrier and Morse (1988) note that 300,000 to 350,000 yearling lake trout are stocked each year. The brood stock origin is traced to southern waters of Lake Superior or Isle Royale. Stocking sites are dispersed along the entire Minnesota shore, sites that were frequented by spawning lake trout in pre-lamprey days.

Smith et al. (1990a) note that in recent years increased emphasis was placed on production of advanced juvenile striped bass as a component of stock restoration programs in the Chesapeake Bay and along the Gulf coast. The decision to stock larger fish is based on a cost/benefit ratio. In each case a decision is made as to whether it is more cost-effective to produce larger, but fewer, juveniles with a higher survival rate or a larger number of smaller fish that have a lower survival rate. In coastal stockings, benefits from stocking larger fish appear greater because of high predatory diversity in marine water. In addition, all hatchery fish are tagged before release. Smith et al. (1990a) describe the procedures for producing larger striped bass for stocking, including: pond design, feeding, mortality estimation, stocking densities, growth, survival, water quality, and management.

3.3.2.3.3 Assessment of Survival and Reproductive Success of Stocked Species

Restoration of fish populations often involves the assessment of both survival and reproductive success of stocked fish (Marsden et al., 1989). To compare the relative survival of strains of lake trout stocked into Lake Ontario, strains were marked by the management agencies with either unique fin clips, coded wire tags, or both (Elrod and Schneider, 1986). Recaptures of marked fish indicate that lake trout strains differ in their survival after stocking (Schneider et al., 1983).

Assessment of reproductive success is not as straightforward as assessment of survival, especially if more than one hatchery strain uses a single spawning area. The comparison of reproductive success among strains requires identification of the parents of naturally produced young. Genetic markers (e.g., allozymes and mitochondrial DNA) can be used to identify the parental stocks of young lake trout produced in the wild.

On-going research on the restoration of a self-perpetuating spawning population of lake trout to Lake Ontario includes attempts to determine which of the genetic strains introduced to the lake are reproducing (Marsden and Krueger, 1989). Management decisions related to the restoration of a self-perpetuating population of lake trout in Lake Ontario would be improved with information about differences in reproductive success of stocked strains. Assessment of differential reproductive success of naturally spawning mixed-stock fish populations requires the use of genetic markers that are transmitted between generations. Marsden and Krueger (1989) described the identification of hatchery lake trout strains that successfully reproduced on a single reef in Lake Ontario in 1986. The analysis used allozyme data from parental stocks and naturally-produced young and represents a novel application of the maximum-likelihood method of mixed stocked analysis. Lake trout fry captured in 1986 were estimated to have been produced by Seneca X Seneca strain (78%) and by Seneca X Superior crosses (20%). Eight other strains and strain hybrids were estimated to be absent from this population of young fish. Similar results, although with different proportions, were found for four hatchery year classes that were prorogated from gametes taken from adult lake trout captured in the eastern basin of Lake Ontario. Before these analyses are used to develop future stocking programs, the reproductive success of strains must be assessed over several years and spawning locations.

3.3.2.3.4 Stocking as an Enhancement Tool for Marine Species

MacCall (1989) reviewed the status of marine fish hatcheries and concluded that they have a long history of expensive operation with no demonstrable positive effect on the resource. In particular, he noted that although cod larvae were released into the Atlantic for nearly a hundred years (i.e., 50 billion larvae between 1890 and 1950) the operation was terminated in 1952 because of the lack of demonstrable beneficial effects on the fish population (Duncan and Meehan, 1954). A similar hatchery operation for Norwegian cod also failed to demonstrate positive results over an extended time period (Solemdal et al., 1984).

MacCall (1989) noted that it is difficult to detect the survival rate of hatchery-produced fish. Full hatchery operation may be necessary to determine effectiveness of a program. Although expensive, modern techniques of genetic marking and fingerprinting provide new tools for determining hatchery success. Such tools should allow identification of hatchery-produced fish in subsequent catches and identification of the genes of the hatchery-produced fish in later wild generations. Development of a genetic strain requires a long time and large investment in hatchery facilities before the program's effectiveness can be evaluated. Thus, although determining the success of a marine hatchery program is now feasible, it remains extremely difficult and expensive.

MacCall (1989) concluded that the few cases where marine hatcheries seem to have produced recoverable fish have been associated with estuarine (see below) rather than open-ocean fisheries. He cautions that effective management of fisheries on declining natural stocks has always been difficult to obtain.

Marine fish hatcheries may be a functional and productive restoration or enhancement technique for the few species that have accessible spawning aggregations, culturable embryonic and larval stages, and adaptable juvenile rearing stages. In addition, fishery demand for these species must justify continual large amounts of capital and operational funding (Buckley, 1989). So far, all of these factors are satisfied for only one species in the United States, red drum (Rutledge and Grant, 1989).

Red drum, an important sport and commercial finfish, began dramatically declining in Texas in the 1970s, prompting regulatory measures such as size, bag, and possession limits, restrictions on gill nets and their operation, a commercial quota, and license restrictions. These steps proved ineffective and, in 1981, the commercial sale of red drum was banned. In 1982, the Texas Parks and Wildlife Department began operating a marine fish hatchery to enhance dwindling populations of red drum. By 1989, over 42 million red drum fingerlings (25 mm TL) had been raised and stocked in Texas bays. Recent expansions of the facility have boosted production capabilities to 20 million fingerlings annually. Impact evaluation began in 1983 and indicate that the enhancement of red drum populations in Texas bays by stocking is a successful and effective management tool.

Matlock (1987) postulated that larval recruitment of red drum into bays could be a limiting factor of annual year class abundance. Matlock's limited recruitment theory and the possible use of hatchery-produced fish for stocking was tested in St. Charles Bay from 1979 to 1981. Matlock found a significantly higher mean catch of red drum in bag seines following stocking compared to an adjacent bay that was not stocked. The success of the pilot study lead to the development of a fish hatchery supported by the Gulf Coast Conservation Association, Central Power and Light, and the Texas Parks and Wildlife Department.

The success of enhancing red drum populations using hatchery-reared fingerlings has been evaluated since 1983. Hammerschmidt and Saul (1984) reported that overall 24-hour survival of stocked fish held in cages was 86.2% (i.e., mortality associated with harvest and hauling stress). Dailey and McEachron (1986) captured stocked red drum fingerlings in San Antonio Bay up to one and a half months after stocking, after which they were not vulnerable to capture gear. Mean catch rates in gill nets in the Corpus Christi Bay system were much higher in the two years after stocking than in the years before stocking. The size of the fish caught in each subsequent year reflected the recruitment of stocked fish when compared with unstocked systems. Stocking also apparently increased the fishing success of sport-anglers for red drum. The mean landing rate by fishermen increased 150% in the stocked bay system, but only 50% in the unstocked bay over the mean historic rate. A cost-benefit analysis demonstrated the economic viability of a salt water hatchery even at low survival rates.

The authors suggest several results that are generally applicable to the evaluation of other marine stocking programs:

- The effectiveness of marine stock enhancement programs cannot be evaluated on an *a priori* basis. To measure the impact, fish must be stocked. Once they have been stocked successfully, the system will be forever changed;

- Substantive impact on a large, dynamic fishery may require massive stockings. Experimental designs using small numbers of fish may not show up against annual variation in population abundance; and

- Although managers may strive for statistical accuracy measured with a micrometer, benefits may only be measurable with a yardstick. Long-term trends may be the only indicator of success.

Willis and Roberts (1991) note that the Florida Marine Research Institute Stock Enhancement Research Facility was fully operational since 1988. The goal of enhancement efforts is to restore a severely-depleted or extirpated stock so that natural reproduction and recruitment can successfully occur. Interest in enhancement of marine fish stocks by release of hatchery-produced fish is at an all time high. However, there is a paucity of scientific information concerning many aspects of enhancement. To approach stock enhancement in a meaningful manner and prior to initiation of fish releases, the Institute solicited scientific opinions from experts in the fields of population genetics, fish disease, and fish and hatchery management. A draft policy was formulated that identifies permitting procedures and requirements for collection of broodstock. The policy also delineates genetic, health certification, disease analysis, and mark/recapture procedures for fish released into the public marine waters of the state. The success of the program in the actual restoration of marine fish stocks will be evaluated in the future.

3.3.2.4 Habitat Restoration and Replacement

3.3.2.4.1 Restored Estuarine Wetlands and Intertidal Habitat

Some species of anadromous salmon utilize wetlands as juveniles when migrating to the sea. Wetlands are believed to be important to provide habitat for temporary residence, seawater acclimation, refuge from predation, and optimal foraging conditions (Shreffler et al., 1992; Simenstad and Thom, 1992). Native wetland habitat in the Puyallup River has virtually disappeared (i.e., 98.6% destroyed) through dredging, diking, and filling. In 1985-86 a 3.9-ha wetland was constructed in the tidally influenced portion of the Puyallup River to replace a 3.9-ha wetland 1.6 km downstream which was filled for development (Shreffler et al., 1990). The wetland was designed to have 50% of its habitat area support juvenile salmon. The wetland system is unique in size, location, and design and is currently the largest estuarine mitigation project in the state of Washington (Cooper, 1987).

The recently restored wetland was studied for usage by out-migraters (Shreffler et al., 1990). Based on mark recapture studies <1% of the out-migrating salmon entered the wetland. Residence for juvenile salmon averaged between 2-38 days, dependent on species. Salmon did forage and grow while resident in the restored wetland (Shreffler et al., 1992). Foraging was highly selective on detritivores and the authors speculate that a beneficial detritous-based food chain is developing in the wetland. The system is still considered to be undergoing colonization. Usage level, food consumption rates, growth and survival could not be compared with either other natural systems or with seaward migrants who did not use a wetland. Interpretation of the system's status as a nursery area is difficult because of insufficient data and the early stage of the system's colonization.

Brun et al. (1991) described a plan for mitigating loss of crab and salmonid habitat following construction planned for Grays Harbor, Washington. The design called for placement of oyster shell for crabs and construction of a new coastal slough for salmonids. Both dungeness crab losses due to dredging and shallow water salmon habitat losses due to a wider and deeper channel would result from channel improvements needed for navigation (Brun, 1991). An estimate of the number of crabs entrained and killed during the improvement project was provided by a mortality model (Armstrong et al., 1987). Juvenile salmon would be affected by the loss of shallow subtidal habitat (1.8 acres) used during out-migration to smoltify.

The following adjustments were made to minimize impacts to crabs and mitigate effects:

- Schedule dredging to avoid times and areas of high crab densities;
- Locate offshore disposal sites to avoid high concentrations of crabs and interference with the fishery;
- Use clamshell dredges where possible to avoid entraining crabs; and
- Provide oyster shell habitat for juvenile crabs in portions of Grays Harbor. This method increases the density of young of the year crabs.

The project would result in an estimated crab loss of 281,000 harvestable crabs. The oyster shell will be distributed and monitored for an 11 year period after initial construction to evaluate the effectiveness of the shell as crab habitat, location of the mitigation site, and stability of the shell plots. After each year of monitoring, the plan will be evaluated. The goal is to provide sufficient high-quality juvenile habitat to compensate for the loss of adults resulting from initial start-up construction and continued maintenance of the shipping channel.

The salmonid plan calls for the construction of four acres of intertidal habitat, replacement habitat considered to be of equivalent biological value relative to that lost. Twenty in-water structures are also proposed to provide shelter for juvenile salmonids.

Brun et al. (1991) report discuss progress to date. Full scale placement of oyster shell was delayed. However, test plots have indicated that oyster shell placed on mudflats has proved effective as habitat for young crabs. A new coastal slough was constructed for salmonid use. Although it is anticipated that it will provide beneficial habitat for out-migrating salmon, results are currently unavailable. The progress of this project should be monitored since it will provide valuable insights into the viability of the proposed restoration techniques. Well-planned, large-scale studies of this kind are relatively rare in the available restoration literature.

Sargent and Carlson (1987) noted that the functional assessment of restored and natural wetland habitat is one of the most important tasks facing estuarine scientists and managers today. Knowledge of the carrying capacity of estuarine habitats and the habitat requirements of all life stages of economically important fish species is critical in making decisions about habitat preservation (Sargent and Carlson, 1987). The success of restored and created wetlands must be judged by functional attributes rather than plant survival. Since fish are an important biological flux mechanism coupling wetlands to estuarine foodwebs, techniques are needed to measure the absolute or relative densities of wetland fish species.

Many techniques are used to sample fish in marsh habitat. Such techniques were grouped into two types by Sargent and Carlson (1987), active gear, such as seines, throw traps, popnets, pullnets, drop nets, block net, and rotenone and cast nets or passive gear such as fyke nets, heart traps, minnow traps, flumes, breeder traps, and gill nets. Active gear types are potentially quantitative on an areal basis but are more effective in areas of open water or limited vegetation. Those used in thick undergrowth are destructive to habitat. Passive gears are not as destructive or labor intensive but cannot be used to estimate absolute densities. Since they are stationary, they collect actively foraging species but under represent the predators and, therefore, have overall greater bias than active techniques.

The plastic fish trap designed by Breeder (1960) appears to be the single method that can be used to compare densities among different marshes or different zones of the same marsh (Sargent and Carlson, 1987). This method performs efficiently in even the densest vegetation while causing only minimal habitat disturbance. Statistical validity through replication, as well as reasonable cost and effort, was demonstrated. The authors conclude that Breeder traps may be useful for functional assessment of restored and newly-created marshes.

3.3.2.4.2 Restored Intragravel Spawning Habitat for Salmonids

Mih and Bailey (1981) described the construction of a machine for the restoration of stream gravel for spawning and rearing of salmon. The machine was designed to remove large amounts of silt and fine sand in the intragravel spaces of spawning streams. The presence of the silt and sand reduces the interstitial flow and oxygen supply, resulting in high mortality of eggs and alevins. Field tests of the machine were conducted in Idaho and Washington. The machine was capable of cleaning gravel to a depth of 15 to 30 cm, depending on the stream gravel condition and speed of the machine. The concentrated silt was ejected onto the streambank but could be discharged into a dump truck and permanently removed from the vicinity. No mention was made of the effect of the removal of silt and sand on subsequent egg and aelvin mortality for the streams tested. Similar machines are routinely used for cleaning gravel in spawning channels in British Columbia, Canada (Helton, 1993).

3.3.2.4.3 Restored Spawning Habitat

Sinning and Andrew (1979) noted that some possible reasons for the decline in the Colorado River basin's Colorado squawfish are water diversion and dam construction which modified physical habitat and temperatures, as well as the introduction of exotic species which compete with or prey on larval squawfish. During low summer flows, water withdrawals have significantly reduced flows in uncontrolled streams. Where streams are controlled, irrigation releases during summer low flows have mitigated the withdrawals in some stream reaches, but the impoundment of high spring flows has reduced sediment flushing. The result in both controlled and uncontrolled stream types was a reduction in shallow areas with reduced flows during the larval rearing period (i.e., reduced rearing habitat). Construction of additional backwater habitat suitable for rearing Colorado squawfish was attempted as a habitat enhancement feature (Sinning and Andrew, 1979). Since squawfish larvae are found in relatively shallow backwaters, which are largely dry during late fall and winter low flows, these parameters were duplicated in the artificial backwaters. The natural backwaters are usually open to the main channel sufficiently that a small amount of water circulation prevents stagnation. Although they have irregular bottom profiles, it was felt desirable to construct the artificial backwater with a regular, U-shaped profile to allow seining or block netting as needed. Percolation of water through the upstream end was designed into the upstream-end dam between the main channel and backwater. After construction of the backwaters, actual squawfish rearing was conducted by the Colorado Division of Wildlife. The outcome of the project was not described.

3.3.2.4.4 Liming to Restore Habitat, Reduce Acidity, and Enhance Fish or Restore Fish Populations

Acidification is currently considered the most serious environmental problem in Norwegian freshwaters. Barlup et al. (1989) described the liming of a chronically acidic Lake and adjoining pond in southern Norway. The area was originally limed in 1981 and then stocked with brown trout at low (Lake Store Hovvatn) and high densities (Pollen Pond). After six years of reacidification, the locations were relimed in 1987. Growth depression during the reacidification process was observed in the lake despite the low density of fish and the superabundance of food. Three months after reliming, a substantial growth response was found in the trout from the lake and mean annual length increment was 68% higher than that of the preceding year. Reliming had no apparent effect on the pond. The results show that the growth response to reliming depends on population density and food availability and suggest that the food conversion rate of the trout is negatively affected in acid waters.

Watt (1986) noted that there are 60 rivers flowing through the southern upland area of Nova Scotia that have the physical potential to support Atlantic salmon stocks. Long range atmospheric transport and deposition of H_2SO_4 has caused the pH level in many of these streams to decline to the point where their Atlantic salmon stocks have been destroyed or diminished. Based on Watt's analysis, the total annual salmon and grills production from the southern upland is presently about 22,700 fish yr^{-1}. The estimate for total production potential in the absence of acidification is about 45,200 fish yr^{-1} for 20.8 km^2 of available salmon habitat.

The pH of salmon streams can be adjusted to satisfactory levels (pH above 5.0) by liming, but fresh limestone must be added at least annually. The total estimated cost for a 20-year project of deacidifying the Atlantic salmon habitat for the area is $95,000,000 (1984 $ Can.). This cost includes the capital cost for road construction, silo construction and replacement, annual lime spreading, silo operations costs, and the costs of a modest monitoring program. The aim of the production effort would be to return the Atlantic salmon production level to the pre-acidification level of 45,200 adults per year. The actual Canadian catch would be about 24,000 salmon, an enhancement of 12,000 salmon above the present average catch. The costs amount to about $400 per restored salmon. The value per landed salmon to the eastern Canadian economy is, on average, less than $100 per fish, hence the liming operation cannot be justified on economic grounds.

3.3.2.4.5 Groundwater Additions to Reduce Acidity in Streams

Effects of acid rain on aquatic communities may be temporarily mitigated by chemical neutralization techniques (for review, see Fraser and Britt, 1982). Mitigating stream acidification presents a special problem because water must be neutralized on a continuous basis, at least seasonally. Zurbach (1984) found that many mitigation techniques used in streams were inefficient, short-lived, and expensive. Despite these problems, neutralization of acidity in lakes and streams remains the major approach to preserving or restoring aquatic life in poorly buffered waters (Schreiber and Britt, 1987).

Early attempts at neutralizing running water with limestone were ineffective due to inadequate contact time and coating of limestone surfaces. More effective systems have used fine particle sizes such as 2-mm limestone in suspension (Abrahamsen and Matzow, 1984). Other devices have pulverized larger particles *in situ* using diversion wells (Sverdrup et al., 1981) or rotating drums (Zurbuch, 1984). These systems, however, required periodic refilling.

A technique used by Gagen et al. (1989) in southwestern Pennsylvania to protect stocked trout utilized pumped alkaline groundwater. Groundwater pumping is similar to natural stream neutralization, because groundwater inflow is the principal agent responsible for buffering acidity in many headwater streams (Sharpe et al., 1984; DeWalle et al., 1987; Peters and Driscoll, 1987). Furthermore, the deleterious sedimentation possible with limestone addition is avoided.

Gagen et al. (1989) reported that groundwater addition during the springs of 1985 and 1986 increased mean stream pH from 4.9, upstream of 3 wells, to 6.0 in the treatment section. It also reduced dissolved aluminum. The combined effect of increased pH and decreased aluminum concentration detoxified the stream, as has been reported for other successful liming projects (e.g., Rosseland and Skogheim, 1984; Rosseland et al., 1986). No mortality of caged trout occurred in the treatment section of the stream except during a large runoff event that overwhelmed the capacity of the wells to neutralize the stream segment. However, mortality was rapid for caged trout upstream of the wells, occurring by 67 hr for brook trout and 29 hr for brown trout. Pumping alkaline groundwater provided a relatively inexpensive alternative to limestone addition. Annual operating costs to maintain a trout fishery from April to October were estimated to be $1,500.

3.3.2.4.6 Restored Water Quality

Hawkins et al. (1992) described a technique for restoring water quality to highly eutrophic dock regions in inner city areas. Poor water quality in such regions is primarily a result of chronic contamination by nutrient input from sewage. As noted by Hawkins et al. (1992), port development in the British Isles led to extensive systems of enclosed dock basins along major estuaries. Many have recently fallen into decline or total disuse and are the focus of ambitious restoration schemes. Hawkins et al. (1992) broadly appraised the state of water quality in disused docks on a nationwide basis in the United Kingdom and identified major problems. Problems are mainly related to the eutrophic, polluted nature of source waters. Anoxic bottom waters are common in summer when stratification occurs. Unsightly dense phytoplankton blooms are also a major problem. Two case studies were examined in detail.

In one high-salinity Liverpool dock, Sandon, an experimental fish farm was run between 1978 and 1983. Dramatic water quality improvements occurred that were attributed to the combination of artificial mixing using an aerator device and dense populations of naturally settled and cultivated mussels acting as a giant biological filter. Anoxic bottom waters were eliminated and the water became much clearer. A diverse benthic community dominated by mussels developed and fish proliferated. This work prompted a more detailed study of the effectiveness of mixing and biological filtration by mussels in the nearby South Docks complex, which is part of an urban renewal scheme. The effectiveness of artificial mixing combined with use of mussels as a biological filter was confirmed. Improvements in water quality were also noted in one enclosed dock due to the filtering action of natural settled mussels. A diverse mussel-dominated community also developed in the South Docks.

Water quality problems are more intractable at low salinity docks at the head of the estuaries. Flushing of docks is only partially effective. The benthic community is impoverished with no natural candidates for use as biological filters. A diverse fish community does exist with potential for a recreational fishery. Water catchment cleanup is the proposed strategic solution. Other suggested approaches included, isolation, followed by installation of mixing devices and a biological filter, chemical methods to strip nutrients and reduce phytoplankton, and speculative biomanipulative approaches.

Hawkins et al. (1992) concluded that restored disused docks are valuable for water-based recreation, research and education, and promoting tourism and redevelopment in urban areas. Aquaculture is less likely to be successful. Restored dock systems are considered invaluable in urban conservation. They compensate for destroyed saline lagoons and promote wildlife and fisheries.

Nordby and Zedler (1991) noted that changes in the assemblages of fishes, bivalves, and polychaetes were evaluated in relation to wastewater inflows at Tijuana Estuary, and impounded streamflows and mouth closure at Los Penasquitos Lagoon. Freshwater from sewage discharges or winter rains lowered water salinities and had major impacts on channel organisms of both southern California wetlands. Benthic infaunal assemblages responded more rapidly to reduced salinity than did fishes with continued salinity reduction leading to extirpation of most species. Both the fish and benthic invertebrates became dominated by species with early ages of maturity and protracted spawning seasons. Between-system comparisons showed that good tidal flushing reduced negative impacts on both the fish and benthic assemblages. Recovery of these systems would require elimination of the man-made disturbances and time for native species to reinvade from refuges within the region's coastal water bodies.

Future plans in San Diego County to discharge treated wastewater into coastal streams are predicted to cause shifts from wetland salt marsh vegetation to brackish marsh species. Based on their results, Nordby and Zedler's (1991) now predict that the resulting changes would impact fish and macroinvertebrate populations, perhaps causing extermination of most or all of the existing channel biota.

Livingston (1985) described the partial natural recovery of fish communities following water quality restoration. Shallow coastal portions of the northeast Gulf of Mexico have high seasonal variations in variables such as temperature, salinity, and nutrient distribution. A nine-year comparison of a polluted and a non-polluted estuary was carried out to determine fish distributions in relation to known trophic states and habitat characteristics. In the unpolluted habitat, the fish community was resilient to extreme changes in the natural environment. The relative abundance and general feeding pattern of dominant fishes remained stable from year to year. In the polluted system, high natural habitat variability was superimposed over water quality changes due to pulp mill effluents. Mill discharge caused increased color, turbidity, and nutrients and decreased oxygen relative to the natural system. The altered habitat was associated with reduced benthic macrophytes and lower fish abundance. Grassbed species were replaced by plankton-feeding fish and seasonal patterns of dominance were altered. Partial recovery of fish assemblages followed water quality restoration with a shift in the pattern of dominance toward the unaffected estuary. Alteration of the benthic macrophytes appeared to be a factor in the response of the fish community. The results suggest that with time and elimination of the mill effluents the benthos would recover, followed by reestablishment of the grassbed fishes. No estimate was given for the time to complete recovery.

3.3.2.4.7 Restored Water Quality after Dam Construction

Ward et al. (1979) suggested a series of ameliorative measures to protect the biotic communities of modified downstream lotic systems following daming and impoundment:

- Dams constructed to allow water to be drawn from a varying combination of reservoir depths would enable simulation of the natural daily and seasonal thermal patterns characterizing a given stream reach;
- Removal of sediments may be accomplished by releasing high water flows in a pattern simulating the natural flow regime and, thus, retaining a more natural receiving stream environment;
- Air drafts installed in release valves will normally alleviate oxygen deficits that may otherwise occur in the receiving stream;
- Discharge-way deflectors offer promise as a means of reducing gas supersaturation levels resulting form water falling from high dams mixing with air that is subsequently dissolved under the hydrostatic pressures in deep-plunge basins; and
- Screening turbine intakes, constructing fish ladders and trucking adults and juveniles around dams have been used to preserve andromous fisheries.

In the event of an oil discharge or other discharge damaging downstream fish populations, initiating the above measures, if not already in place, would hasten recovery.

3.3.2.5 Habitat Enhancement

3.3.2.5.1 Artificial Reefs (benthic and semi-pelagic fish) and Fish Aggregating Devices (pelagic fish)

Duedall and Champ (1991) and Sheehy and Vick (1992) reviewed marine artificial reef programs. A major scientific question for the use of artificial reefs in restoring fish populations is whether reefs lead to increased overall fish production or merely provide for redistribution (via attraction) of the existing population.

Most U.S. coastal states have very active marine artificial reef programs, spending millions of dollars to develop reefs for use by sport and commercial fishermen and recreational divers. Japan, the world leader in reef design, has spent billions of dollars developing, engineering and deploying new designs. Reefs are constructed of a wide variety of materials including rubble, discarded wastes, junked automobiles, aircraft, boxcars, quarry rock, and marine-grade concrete cast in large, specially-designed reef units. Ideally, artificial reefs should be made of economical materials that are placed on the seabed or prefabricated on land in a design that will serve the specific purpose of attracting fish.

Artificial reefs are designed not only to support general or specific fisheries, leading to the creation of new fishing grounds, but also to increase the production and diversity of colonizing organisms. Reefs may rebuild fishery stocks, or mitigate some of the impacts or losses related to coastal development. In California, work on artificial reefs was supported by a power company (S. California Edison) to explore the potential for reefs to mitigate the effects of power plants on coastal areas.

Fish abundances at and near an artificial reef are always greater than abundances in nearby sandy areas. Generally, larger and more complex structures attract more species and greater numbers of different fishes. In American Samoa, the fish catch-per-unit-effort (lbs. per vessel) was 8.4 to 17.4 for a control area, 40.4 to 49.8 at an artificial reef, and 52.8 to 90.9 for an offshore bank. Sampling limitations make it difficult to accurately determine the origin of the fish found at a reef and the area's overall capacity to support fish production. Some reefs (in Japan) are designed not necessarily to attract or produce fish directly, but rather to induce turbulence or upwelling that stimulates primary productivity and, thus, enhances the entire system.

Duedall and Champ (1991) feel that it is too early to determine whether enhanced reef fish harvest results from a net increase in fish production, redistribution of stock, or some combination of the two. The answer to these questions are critical in determining the role of artificial reefs in fish restoration projects. Restoration rarely has as its goal the return of fish to a given area at the expense of adjacent regions.

Buckley (1989) is highly critical of the fact that considerable artificial reef construction has been in response to incentives for solid waste disposal. Recruitment and survival of juvenile fish is restricted because of the use of "materials of opportunity" for constructing artificial reefs. Fish aggregating devices (FADs) are often lost (buried by sand or moved by waves and current) as a result of inadequate design and engineering.

Buckley (1989) noted that altering marine habitats to increase fishery productivity is well within current technological capabilities. The two most common methods of marine habitat alterations, artificial reefs and FADs, can be used to enhance marine fisheries by increasing the amount of marine resources available for harvest and controlling the temporal and spatial distribution of these resources (Buckley and Hueckel, 1985; Wilson and Krenn, 1986; Alevizon, 1988; Buckley et al., 1989; Polovina and Sakai, 1989 and others; see Buckley et al., 1985). He also notes that evidence is mounting that biological development on artificial reefs can also supplement natural production and recruitment of reef-related species. The capital costs for artificial reefs and FADs can be low relative to other enhancement actions and operational costs can be moderate. However, historically attempts to apply these habitat alterations were both effective and ineffective in enhancing marine fisheries. The level of effectiveness appears to be directly correlated with the amount of science included in applying and evaluating these technologies.

Buckley (1989) emphasized that there was a recent evolution toward designing and evaluating artificial reef projects that target specific questions about resource enhancement, particularly recruitment and survival of juveniles. Recent research has shown that, when applied correctly, this technology creates long-term, if not permanent, alterations of benthic habitats, which develop biologically into replicates of productive natural reefs, primarily for benthic and semi-pelagic species (Buckley and Hueckel, 1985; Wilson and Krenn, 1986; Alevizon, 1988). These alterations enhance the aggregation and production of important resources at locations that are atypical of the natural system. Artificial reef technology gives fishery managers some degree of power to direct the marine ecosystem and selected biota toward desired responses. These changes can increase the accessibility and fishability of traditional or new resources and alleviate problems of fishery interaction by redistributing competing fisheries.

The first quantitative assessment comparing the potential for FADs to enhance marine fisheries for pelagic species relative to offshore bank and open-water areas was completed in 1987 (Buckley et al., 1989). This study verified the potential for correctly-sited and engineered FADs to enhance marine fisheries to a level comparable to large, productive, offshore banks.

Buckley (1989) noted that successful application of artificial reef and FAD technologies can only occur if there is adequate funding for research, development, and evaluation of each project. The first criterion must be the careful evaluation of realistic and justified fishery enhancement objectives, which are the bases for habitat alteration. These objectives must address the species to be enhanced, fisheries that will benefit, and potential for adverse impact. In addition, appropriate siting and design criteria must be applied. The physical and spatial designs of artificial reefs must consider habitat configurations that allow replication of natural reef systems, especially for the recruitment, survival, and growth factors that control production.

Unlike other authors (as reviewed above), Buckley (1989) noted that there have been enough good artificial reef programs in recent years to provide ample evidence that this habitat alteration has both production and aggregation (enhancement) functions for the associated biota. He further concludes that the FADs' aggregation capabilities can also result in production through optimizing the use of alternate, atypical food resources. Although the artificial reef idea has merit, most current applications and designs are flawed. This is primarily due to the prevalent use of materials of opportunity to construct the reefs. In closing, Buckley (1989) states that technologies for artificial reefs and FADs have suffered from inadequate, haphazard funding, and lack of realistic, justified fishery enhancement objectives as the incentives for altering habitats. Solving these two major constraints will allow refinement of these technologies and accurate evaluation of these habitat alterations as a basis for enhancing marine fisheries.

Hueckel and Buckley (1982) noted that the successful use of marine habitat enhancement (using artificial reefs) to increase availability of desirable bottomfish to recreational anglers based on the reef developing into replicates of natural rocky reef communities. Such communities have resilient populations of target fishes. These criteria require habitat enhancement sites to be located in areas that maximize the potential for production of organisms found in a balanced rocky reef community. Physical and biological parameters of 26 potential sites were compared to physical parameters and an index of common biota from three rocky reef control sites in the Puget Sound region. Hueckel and Buckley (1982) felt that these comparisons gave the best possible information on each sites' biological production potential. Their findings were used to make recommendations on the sites' potential for enhancement.

Their results indicated that acceptable sites all exhibited:

- Good biological production potential indicated by the presence of rocky reef organisms orientating or attached to existing artifacts;
- Stable bottom substrate; and
- Good water quality and currents.

The unacceptable sites had:

- Silty substrates with inhibited biological production; and/or
- Steep sloping bottoms, which negate placement of habitat enhancement structures.

Hueckel et al. (1983) reported that artificial reefs were subsequently constructed in central to southern Puget Sound to provide recreational fishermen access to productive bottom fishing. Key rocky reef fish (rockfish, lingcod) are habitat-limited in this region. Before site construction, the Washington Department of Fisheries (WDF) reviewed the differences in materials and designs of artificial reefs relative to their ability to attract fish. Turner et al. (1969) observed more fish attracted to concrete shelters when compared with quarry rock, automobiles, and street cars off California. Sheey (1982) noted that fish gathered around concrete blocks piled closely together but not around scattered blocks off the Japanese coast. Sheey also reported the most effective artificial reefs in Japan are constructed to maximize relief and utilize spacing between individual structures.

Based on the above review, the WDF constructed seven habitat enhancement structures from concrete pilings, hollow core slabs, and large rubble off Gedney Island, Puget Sound. These structures have attracted at least eight recreationally important fish species, six of which were not present during pre-construction baseline surveys. Recreational fishermen caught primarily flounder and rockfish amongst the enhancement structures, averaging 3.4 fish per 4.0 hour trip. The flounder were most likely caught over sand bottom and not on the enhancement structures.

The structures were surveyed by SCUBA transect techniques on a monthly basis to enumerate recreationally important fish species. The most abundant species were shiner perch, striped seaperch, pileperch, and rockfish. Lingcod egg masses were observed on the structures indicating that they provided suitable spawning habitat for an important recreational species. Lingcod numbers and their egg masses increased two-fold on the enhancement structures from August 1980 to June 1981. The enhancement structures are, thus, contributing to increased lingcod production.

The authors concluded that the structures provided the necessary vertical relief to attract large numbers of schooling surfperch and crevices necessary to attract sedentary rockfish and lingcod. They felt that future structures should incorporate additional small concrete rubble to provide more protective habitat for small rockfish. Higher relief structures may be required to attract pelagic rockfish species.

Hueckel et al. (1989) noted that the application of artificial reefs as mitigation for injured or lost rocky habitat was not extensively studied. Mitigation projects often fail to achieve their objective of no net-loss of habitat because of using unproven habitat modification techniques with inadequate site selection and project evaluation studies. Some projects fail because they attempt to change the community structure through introduction of a desired species. The species may not survive because it is inappropriately placed or eventually out-competed by a resident species. Other unsuccessful projects result from the introduction of new habitat in physically inappropriate locations.

Hueckel et al. (1989) described the construction of an artificial reef on a featureless sand bottom as mitigation for the man-caused loss of rocky subtidal habitat in Elliott Bay, Puget Sound, by a shoreline development project. It was predicted that the artificial reef would develop a greater number of economically important fish species than the development site. A total of 181,400 metric tons of quarry rock was used to construct fourteen 41 m by 15 m by 6 m reef structures in a 2.83 ha area during May 1982. Species diversity and densities on the mitigation reef surpassed that observed on the rocky bottom of the development site during the first eight months of submergence. Some displacement of resident fish may have occurred, with flounder diversity and density greater on the adjacent sand bottom rather than between the mitigation reef structures. Artificial reefs also caused a decline in benthic infaunal density and diversity in the sand under and around the slabs of 5 and 7 year-old Puget Sound structures. The authors concluded that artificial reefs can be used to compensate for man-caused losses of rocky habitats. The artificial reef developed an assemblage of economically important fish species similar to, but greater than, the impacted habitat.

Three sites in Chesapeake Bay were used to examine the feasibility of utilizing artificial reefs of five types to improve Catch-Per-Unit-Effort (CPUE) of black sea bass, tautog, grey triggerfish and toadfish (Feigenbaum et al., 1989). The reef types tested were concrete igloos, concrete pipe pyramids, high surface area tires, unballasted tire bundles, and tires embedded in concrete. Fish populations on the reefs were evaluated by catch rate using standard two-hook bottom rigs. SCUBA surveys were undertaken to determine structural integrity and movement of reef units. At two sites, reef CPUE's were significantly higher than non-reef, control stations while, at a third, no significant difference was observed. The authors concluded that, in the lower bay and ocean, test sites were successful in attracting fish and providing seasonal habitat for several desirable species. Spawning occurred on reefs and additional reef construction was recommended provided no user conflict with other groups (e.g., menhaden fishery) occurred. On the other hand, mid-bay reefs attracted few adults and angler success was no better than non-reef control sites. The intermediate salinities may not be attractive to the adults of the target species.

Based on the five artificial reef-types constructed, the authors recommended:

- Unballasted tires should not be used for reef structures because they move offsite during storms;

- High surface-area tires (constructed with a reinforced concrete base with imbedded pipes) are durable in protected situations, but are not recommended because the basic steel framework will eventually corrode;

- Tires embedded in concrete are inexpensive (~$8.00) and durable and are recommended for providing low-profile complexity. Similar structures have been used in New York (Zawacki, 1971) and Florida (Unger, 1966) and a modified version was deployed in South Carolina (Bell, 1984);

- Concrete pipe pyramids functioned well and are recommended, providing epoxy is used to hold the pipes together after cable deterioration. The pyramids can be built for about half of the cost of igloos (below); and

- Concrete igloos appear long-lasting, attract fish, and provide a high degree of angler success. These structures were most highly recommended. Even with a high initial cost ($1,200 plus deployment) they are actually quite economical assuming a conservative life span of 50 years ($24 per year). Construction of these units is illustrated in Blair and Feigenbaum (1984).

The Rigs-to-Reef concept provides an alternative to obsolete petroleum production platform removal (McGurrin and Fedler, 1989). More than 4,000 oil and gas production platforms dot the coastal waters of the United States, most in the Gulf of Mexico. Many of these structures are presently abandoned or targeted for abandonment and removal. Removal is generally accomplished by explosives. This explosive removal of obsolete petroleum production structures results in the death or injury of fish, sea turtles, and marine mammals.

The Rigs-to-Reef concept postulates that large-scale artificial reefs from obsolete oil platforms provide excellent fish habitats and provide a cost-effective means of recycling. Although it may be cheaper to leave the structure at the original site (i.e., topple on site or other actions), site-planning and transport may maximize the probability of recreational fishing use and minimize multiple use conflicts. McGurrin and Fedler (1989) describe the movement of a rig and the subsequent use by anglers. In general, there was not much difference between the artificial reef and nearby natural reefs in terms of perceived fishing quality. All fishermen were willing to pay a limited fee for additional site constructions, perhaps in the belief that it would reduce overcrowding on existing sites and provide more and better fishing in the future. The Rigs-to-Reef projects require extensive funding for construction, maintenance, and management and there is a pressing need to justify the investment.

Relini and Relini (1989) indicated that artificial reefs played an important part in the restoration of inshore biological resources in the Ligurian Sea in regions affected by illegal trawling, incorrectly repaired sandy shores with silty materials, and pollution (mainly sewage discharge). Large wooden structures composed of recycled barges and dock gates were effective in promoting the settlement of organisms, in attracting fish, and in preventing illegal trawling activity in shallow waters. The authors provided only a qualitative assessment of the success of the structures.

Brock and Norris (1987) describe the colonization of an artificial reef specifically designed and configured to support fish. They note that such planned artificial reefs are an integral part of fisheries development and restoration programs in the Far East, but that documentation of fish recruitment patterns to these reefs are scarce. Their study follows the recruitment of fishes to an open-framework concrete-cube artificial reef deployed in 20 m of water in Hawaii. They found that colonization was rapid and that the standing crop on the artificial reef (~2000 g/m^2) far exceeded that of productive natural reefs (~ 200 g/m^2). These data suggested that colonization and turnover are initially high but should stabilize with time and the design of the reef was appropriate for Hawaiian inshore fisheries improvement.

Polovina (1989) examined the potential for artificial reefs to increase fish stocks of marine resources. He concluded that although they are excellent fish aggregators they do not effectively increase standing stock. His conclusion is based on the fact that, although Japan has covered 9.3% of the ocean bottom from shore to 200 m with 6443 artificial reefs ($4.2 billion) from 1976-1987, there was no measurable increase in coastal fishing landings. He also noted that low reef habitat may be lost as a result of high reef construction. The low reef habitat is important to various life stages of fish. For example, the juvenile habitat of the very valuable deep-water snappers was found to be low-relief, flat-bottomed, sandy habitat, previously considered a biological desert. Large-scale construction of artificial reefs would have attracted shallow-water reef fishes at the cost of destroying juvenile habitat for the more commercially valuable deep-water species. Furthermore, Polovina (1989) notes that the limiting factor for most reef fish appears to be recruitment from the larval phase and not available habitat, which although necessary is not limiting. Polovina (1989) concludes that artificial reefs are not a solution to overfishing. Artificial reefs are popular as management actions because they concentrate fish resulting in higher catches initially. Artificial reefs may ultimately prove detrimental to a fishery since they delay the imposing of size limits and quotas.

Bell et al. (1989) noted that South Carolina's state-maintained Marine Artificial Reef Program has begun evaluating manufactured artificial reef structures for consideration in future construction efforts on the state's offshore artificial reefs. Manufactured reef units may become a viable replacement for, or supplement to, many forms of scrap materials currently being used to construct artificial reefs. Designed reef structures made of steel, concrete, or plastic, are readily available through established private industries and offer numerous advantages to fisheries managers attempting to use artificial reefs as effective fisheries enhancement tools. To assess the usefulness of designed reef materials currently available, eight types of manufactured reef units were placed in two artificial reefs off South Carolina. The first reef unit design, consisting of low profile concrete pipes, was deployed in 1985. Additional designs, made from molded plastic domes, were added in 1986. The remainder, consisting of steel cubes as well as other concrete designs, were placed on station in 1987. Each design was evaluated based on its procurement, handling, and transportation costs, as well as its stability, durability, and biological effectiveness.

Construction costs of test reefs ranged from $81/m^3 for the steel cubes, to $168/m^3 for one of the concrete pipe modifications, with a mean cost of $110/m^3. Initial in-water evaluation has revealed severe stability problems with two designs, but detected no structural weaknesses in any of the unit types. Preliminary examinations indicated no measurable differences in established populations of target fish species on the different unit types examined. Two years of observations of the initial concrete pipe design are encouraging, and, at this time, these units appear to offer a viable option for a practical manufactured reef structure for South Carolina's Marine Artificial Reef Program. Assessment of the overall effectiveness of each design will be made through continued monitoring of each test reef over the next two to three years.

The authors indicated that quantitative assessment of fish species was beyond the scope of their study. However, fish censusing will be part of future routine evaluations. Many target fish species were encountered on test reefs throughout the course of the study. The steel-reinforced concrete units had the greatest species diversity. Compared with observations on scrap material reefs of comparable size and age, the biological effectiveness of these reef units appears to be well above average. However, valid quantitative assessments of both scrap and designed reef structures will need to be made before meaningful comparisons can be made.

3.3.2.5.2 Artificial Stream Structures

Koski (1992) reviews restoration of streams by restoring stream structure, stability, currents, pools, and diversity of habitat. Natural materials are preferred (e.g., trees, other wood, boulders). Koski cites several case studies where these methods were effective in enhancing fish populations.

Klassen and Northcote (1988) noted that gabion weirs (i.e., wire cages placed into the stream bed and filled with rocks) appear to be useful tools for the restoration of streams injured by logging. Such streams are subject to debris torrents (i.e., mass movement of soil, rock and wood) as well as reduced dissolved oxygen and/or water flow rates. Consequently, this restoration technique would be appropriate for impacts resulting in low dissolved oxygen, reduced flow, and/or structural injury. These factors contribute to suppressed egg-to fry survival of salmonid species. Gabions were successful in stabilizing spawning areas (Klassen and Northcote, 1986) and creating spawning habitat by improving the intragravel environment.

Klassen and Northcote (1988) described the use of tandem weirs at three sites in Sachs Creek, British Columbia, to improve spawning habitat for pink salmon. The results of Klassen and Northcote's (1988) study suggest that the intragravel environment of injured streams can be restored by gabions within a year. Although egg survival at the one site examined was similar to that of reference sites in the first year, reduction in gravel scour after an initial period of gabion ablishment should improve future egg survival. To ensure full use of the production potential, gabions should be placed in reaches having high spawner densities. Additional benefits of gabions, including improvements in juvenile salmonid rearing habitat and juvenile densities (House and Boehne, 1985; Klassen and Northcote, 1986), help offset construction costs. At Sachs Creek, construction costs per site decreased with experience of installation (from $5,244 to 3,827 to 2,985, in that order, in 1982 Canadian dollars). Additional costs of gabion maintenance are becoming evident 4.5 years after construction since ruptures have developed in three of the gabions. The successful use of this technique is currently being questioned (see below).

In recent years an increasing share of fishery management resources in the western U.S. have been committed to alteration of fish habitat with artificial structures such as log wiers or gabions (Frissell and Nawa, 1992). The authors caution that large and costly projects continue to be planned and implemented by federal and state agencies with little or no analysis of their effectiveness. The few evaluations of artificial-structure projects in the Pacific Northwest have shown mixed results. Hall and Baker (1982) and Hamilton (1989) summarized published and unpublished evaluations of the effectiveness of fish habitat modification projects in streams. Although studies of apparently successful projects (e.g., Ward and Slaney, 1981; House and Boehne, 1986; Klassen and Northcote, 1988) were widely cited, studies with less favorable (neutral or negative) biological effects have been published less frequently.

Several studies have indicated that structural modifications can be ineffective or damaging (Frissell and Nawa, 1992). Hamilton (1989) observed reduced trout abundance in a northern California stream reach with artificial boulders compared with an adjacent unaltered reach. A large-scale habitat modification program in Fish Creek, Oregon, produced cost-effective increases in fish production from opening of off-channel ponds, but generally negative or neutral effects from boulder berms and log structures. In Idaho, the Department of Fish and Game found little evidence that in-stream structures increased abundance of juvenile chinook salmon and steelhead and in one project more than 20% of the structures failed during their first winter. In Utah, Platts and Nelson (1985) found that outside a fenced enclosure, artificial structures were destroyed by livestock and grazing-related streambank erosion. Babcock (1986) noted that 75% of the structures in a Colorado project failed or were rendered ineffective by a flood two years after construction. Several of the remaining structures apparently created migration barriers for fishes, a problem also observed in Oregon (Frissell and Nawa, 1992).

For artificial structures to function successfully, they must meet carefully-defined objectives specific to target species, life history stage, and prevailing physical conditions (Everest and Sedell, 1984). The design of such structures must be closely tailored to geomorphic and hydraulic conditions (Klingeman, 1984). To meet both biological and economic objectives, the gabions and log weirs must remain intact at the installation site for their projected life span (i.e., 20-25 years). In the northwest U.S. most structures have not been in place long enough to assess their durability across a range of stream flows.

Frissell and Nawa (1992) evaluated rates and causes of physical impairment or failure of 161 fish habitat structures in 15 streams in Oregon and Washington following a flood of a magnitude that occurs every 2-10 years. The incidence of failure or functional impairment varied widely among streams. The median failure rate was 18.5% and the median damage rate (i.e., impairment plus failure) was 60%. Damage was frequent in low-gradient stream segments and widespread in streams with signs of recent watershed disturbance, high sediment loads, and unstable channels. Rates of damage were higher in larger and wider streams. Comparison of 5-10 year damage rates from 46 additional projects showed high but variable rates in regions where peak discharge at 10-year recurrence intervals has exceeded 1.0 $m^3\ sec^{-1}\ km^{-2}$.

At numerous sites, structures were judged to cause inadvertent adverse physical effects such as:

- Accelerated bank erosion at log weirs;
- Direct damage to gravel bars and riparian vegetation by heavy equipment;
- Felling of key streamside trees to provide sources of materials, causing loss of shade and bank stability;
- Flood rip-out of riparian trees used to anchor log structures;
- Aggragation of gravel bars or silt and sand deposits which caused shallowing and loss of microhabitat diversity in preexisting natural pools; and
- Torrents of bed load and debris triggered by collapse of structures during the flood.

Eggs and fry of fish that spawned in the gravel above log weirs, as well as juvenile fishes wintering in and near the structures, were possibly killed when the structures failed and washed out. Fragments of epoxy or resin used to anchor structures were very common in many pools and there is evidence that these materials can be toxic to fish. Frayed cables and sheets of ripped out geotextile or chain-link anchoring material at damaged structures created obvious aesthetic liabilities. Furthermore, repairs may have exacerbated initial damage (Frissell and Nawa, 1992). Results suggest that commonly prescribed structural modifications often are inappropriate and counterproductive in streams with high sediment loads, high peak flows, or highly erodible bank materials.

The wide range of failed structures indicates that simple changes in structural design or materials are unlikely to overcome the problem of high damage rates. Structure designs that failed least often were those that minimally modified the preexisting channel, such as cabling intended to stabilize natural accumulations of woody debris. Elaborate log weirs and other artificial structures that cause immediate changes in channel morphology and hydraulics were subject to high rates of damage. At least in the study area, it is unrealistic to expect the installation of new artificial structures to stabilize channels and, in fact, the opposite result may be likely. Within the study area, the stream habitats most important for fish and most in need of restoration are those least amenable to structural modification. Frissell and Nawa (1992) conclude that restoration of fourth-order and larger alluvial valley streams, which have the greatest potential for fish production in the Pacific northwest, will require the reestablishment of natural watershed and riparian processes over the long term. They recommend that restoration programs for their study area follow a hierarchial strategy that emphasizes prevention of slope erosion, channelization and inappropriate floodplain development, especially in previously unimpacted habitat refugia, rehabilitation of failing roads, active landslides, and other sediment sources (logged slopes), and reforestation of floodplains and unstable slopes.

3.3.2.5.3 Impoundments and Spawning Channels

Chabreck et al. (1981) reported that marsh impoundments are constructed for wildlife habitat improvement to restore traditional salinity regimes and to prohibit drainage. The overall effect of such impoundments is to create a stable environment for fish, thus aiding in their return to an area or enhancing their abundance over previous depleted values. The authors compiled a list of the types of impoundments and their relation to fish:

- Permanently flooded freshwater impoundments in coastal marshes provide ideal habitat for freshwater fish when depths are adequate;

- Manipulated freshwater impoundments only provide freshwater fish habitat in canals or deep channels not subject to drying;

- Permanently flooded brackish water impoundments serve as vital nursery area for estuarine fish and may produce organic detritus which serves as a primary food source for estuarine fish. Levee systems may reduce nursery areas; and

- Manipulated brackish water impoundments may also serve as important fish nursery areas and detritus production may actually be increased.

Sanner et al. (1982) described the factors of critical importance in selecting sites for habitat enhancement via spawning channel as:

- Groundwater height;

- Groundwater temperature;

- Groundwater gradient;

- Groundwater chemistry;

- Flooding risk;

- Availability of substrate; and

- Availability of broad stock.

Careful site selection is the most critical factor in the potential use of the site for fish spawning.

3.3.2.5.4 Dredge Spoil Islands

Thompson et al. (1983) described the alterations in the Atchafalaya River Delta leading to a new fish nursery area. The dynamics of coastal Louisiana's fish fauna are influenced by the cycle of growth and decay of river deltas and the accompanying change in hydrologic and salinity regimes. Diversion of Mississippi River water down the Atchafalaya River is forming a new delta and creating wetlands in Atchafalaya Bay. Early reports suggested that as freshwater drained into Atchafalaya Bay, nursery capacity would be lost and the cold water associated with winter and spring floods would depress productivity. Data from Thompson et al.'s (1983) study suggest that the emergence of the delta islands has enhanced nursery capacity. The islands provide protected areas that act as temperature refugia against cold, winter riverine waters, and the shape of the island is correlated with the degree of protection afforded. Those areas receiving the greatest degree of protection had significantly higher total number of fish species and total number of animals.

The authors recognize that the shape of artificial islands developed from dredge spoil may influence nursery potential and, thus, be important to management of fisheries. They recommend that dredge spoils resulting from future navigation channel projects in the Atchafalaya Delta be deposited in an altered morphology that would provide habitat with reduced cold water riverine influence. The fishery resources of the delta need areas of temperature refugia that function as stabilizing factors and lessen the impact of cold river water during the critical time of year when many nekton utilize the delta. Other factors that are generally important in the design of dredge spoil islands (Kennedy et al., 1979) include:

- Size (smaller islands (5 to 25 acres) are likely to have rapid ecological development);

- Configuration (produce a multifaceted an island as practical under local conditions of water current and elevation);

- Substrate (the type of substrate may not be beneficial to all types of biological resources); and

- Elevation (the type of vegetation desired and the biological resource using the area should determine the elevation variation).

3.3.2.5.5 Structural Modifications

Knox et al. (1979) described watershed projects designed to protect or mitigate losses of fish or wildlife and riparian habitats as a result of channel work for flood damage reduction and drainage in Indiana. The Upper Big River Project originally called for enlargement of the lower five miles of channel but was changed to a drift-and-debris-removal project to protect a colony of endangered bats. The authors recommend drift and debris removal as the method of channel improvement that has the least impact on fish and wildlife habitats. Restricted flow rates along five-miles of the Middle Fork Anderson River were also corrected by this method, i.e., removing fallen trees and logjams.

To offset the losses of fishery habitat caused by modifying a flood protection channel, a fishway of pools and riffles was developed. In the years following construction, species diversity was consistently high (i.e., greater than an upstream natural channel) and the channel has become naturally revegetated.

Presently when channelization is planned, the construction activity is conducted from only one side and the route follows existing channel alignment. Where possible, large trees are left standing and protection of vegetation along one bank is accomplished. This has proven to be a valuable tool in preserving integrity of the natural channel, provided source for revegetation, and maintains some of the riparian habitat.

In addition, the majority of completed projects have required rip-rap deflectors with excavated fishpools to compensate for the loss of aquatic habitat. Monitoring has shown that the fishpools are self-maintaining and support populations of game fish. The authors summarize protective and/or enhancement techniques associated with channelization in Indiana as:

- Installation of sediment traps to prevent sediment from leaving a construction site;
- Construction on only one side of a stream channel;
- Removal of waterway obstructions with handtools and small equipment to minimize impacts;
- Construction of continuous pool-riffle fish habitat in bedrock;
- Installation of fishpools with deflectors and constructed riffles in earth sections;
- Woody and herbaceous plantings;

- Maintenance of shade over water;
- Wetland acquisition; and
- Use of fencing and vegetation markers.

The authors concluded the above procedures were useful in protecting and restoring riparian habitats in Indiana.

Burke et al. (1979) noted that dike and other structures used to stabilize banks and develop a navigation channel in the Missouri River eliminated considerable fish and wildlife habitat, and substantially reduced habitat diversity. The transformation of the river into a single channel has eliminated most side channels, islands, backwaters and sloughs that are important feeding, nursery, resting and spawning areas for fish and wildlife. Some structures caused permanent land accretion.

Structure modifications started in 1974 are an attempt to improve conditions for fish and wildlife. Notching structures show promise because the notch helps create small side channels that increase habitat by at least doubling the aquatic edge. Without notches or other types of modifications, land accretion occurs and existing wet areas become permanent land often cleared and cultivated, unusable to aquatic life.

Lowering the height of structures eliminates or slows down land accretion. Sand bars form at such low levels that permanent stands of willows and cotton woods cannot be established. These low structures successfully maintain the navigation channel, while if notched or not attached to the bank they can provide aquatic habitat for use by fish and wildlife.

Burke et al. (1979) noted that it is almost impossible to demonstrate conclusively that the modified structures have improved fish populations because of sampling difficulty. Flathead catfish, freshwater drum, and blue sucker appear to prefer the fast water provided by notches. Shallow sand bars provide nursery areas for young fish and minnows and harvest areas for other species. The deep holes created adjacent to the modified structures provide habitat for fish during low flow periods.

The land accretion process can be stopped by using modified structures (Burke et al., 1979). Diverse habitat can be developed by using a variety of structure designs (i.e., high, low, notched, angled, unattached, and combinations thereof). Conditions for large river fish and wildlife populations at all water levels are, thus, improved. The goal of modified structures was to create a diverse aquatic habitat at various levels without causing further land accretion, permanent water surface losses, bank erosion, or impairing the usefulness of the navigation channel. The techniques described by the authors should prove useful during restoration attempts for fish inhabiting large rivers. Providing new habitat in the event of fish losses should speed recovery, as should replacing injured or destroyed habitat with a functionally similar habitat. Use of the above techniques should be preceded by a pilot study to quantitatively confirm the impression that fish respond to the increased, restored, or improved habitat conditions.

3.3.2.5.6 General Stream Management

The Wisconsin Department of Natural Resources (1967) compiled a document providing guidelines for management of trout stream habitat in Wisconsin. They noted that improving trout habitat in Wisconsin is largely a task of restoration. Although pollution and irrigation was kept under control, much trout water was lost due to dam-building and stream-straightening. Heavy grazing and trampling by cattle and impoundments by beaver have adversely affected streams. In addition, dense canopy of trees and tall brush, shade channels and banks, and reduce in-stream aquatic plants and the understory plants provide essential cover at the stream's edge. Their bulletin deals mainly with measures to improve the channel, the banks and plant life for the welfare of trout. In addition, the authors warn against overmanagement and suggest that more effort should go into preserving untouched streams and their surroundings than into alteration of them.

The authors summarize the main principals in managing trout stream habitat as follows:

- General: Tailor habitat management to the individual stream. This requires thorough examination of the stream and its trout, diagnosis of problems, and a plan for the "cures" before the work is done. Preserve the natural character of streams and their landscapes;

- Health of the stream: Health is defined as the capacity for self-repair. Eliminating dams and protecting stream banks against livestock on some streams are relatively inexpensive measures with great impact in enabling self repair. Encouraging flood control and managing stream bank vegetation are important in allowing the stream to function as trout habitat, but more costly;

- Vegetation: Protection and control of stream bank vegetation are often advisable to maintain favorable trout habitat. The trout-sheltering characteristic of natural channels is enhanced by the right kinds of vegetation, mainly the type that drape into the water. These and beneficial aquatic plants cannot grow well in dense shade of trees and tall bushes. Overshading is an especially acute hazard along small streams. Meadow creeks with low shrubs and grasses appear to have the best all-around combination of productivity and protection. Therefore, woody vegetation should be removed from banks of small streams where groundwater seepage is adequate to keep summer temperatures moderate;

- In-stream alterations: In low gradient streams, keep the water moving. Remove dams and other obstacles to flow (but do not remove meanders). When building in-stream structures, do not impede the current unnecessarily. In high gradient streams, make plunge pools. Pools scoured out by water plunging over large rocks or logs may look turbulent, but near the bottom they are quiet, protected resting places for trout;

- Spawning grounds: To aid spawning, protect and enhance naturally occurring stream bed gravel rather than trying to bring in and deposit new gravel. Experiments in building artificial spawning beds have not yet resulted in a method that meets the requirements of feasibility and of compatibility with the natural landscape; and

- Flood control: Combat floods by reducing overland runoff in the drainage basin above the stream, not solely by reinforcing stream banks.

Many of the above management techniques are viable approaches for restoring and enhancing trout habitat following loss from human activity.

3.3.2.5.7 Fish Passageway Improvement

3.3.2.5.7.1 Ladders, Hoists, Transport Flumes, Trap and Truck

Installation of fish passageways promotes recovery of anadromous fish resources by expanding the area of a stream accessible to spawning and rearing of young (Moffitt et al., 1982). Fish passageway improvements consist of ladders, hoists, transport fumes, trap and truck, step pool structures, and discharge water. In 1967, the state fishing agencies sharing the Connecticut River Basin listed as their goal the restoration of two million shad and 38,000 salmon to the mouth of the Connecticut River (Moffitt et al., 1982). The group determined which mainstem dams needed fish passage facilities, provided design parameters for the proposed facilities, began negotiations with utility companies that owned the dams, and initiated a series of recommendations for a program of salmon restoration based on the release of hatchery reared fry and smolts. Since shad were not eliminated from the lower river, stocking above the dams did not receive priority in restoration.

Fish passage facilities now exist at three dams on the mainstem of the Connecticut River and at two dams on tributaries. Since 1967 Atlantic salmon releases from Canadian and Maine stocks have totaled 689,000 presmolts and 993,000 smolts. Variability in number, strain, and quality of smolts and fry stocked into the river was responsible for fluctuations in the number of fish returning. Until a healthy and abundant Connecticut River brood stock is obtained, variations in returns are anticipated and fish passage serves only as a means for adult capture. Achievement of the population restoration will require that natural production be supplemented with hatchery-produced smolts. Thus, there is a continuing need for some brood stock to be removed from each year's run for the production of eggs for hatchery rearing.

For other anadromous species, at least four to five years are required to detect changes that could be attributed to the initiation of successful upstream or downstream fish passage efforts. Two facilities were operational long enough to provide trend information. At one site, passage of anadromous species (i.e., shad, blueback herring, sea lamprey, Atlantic salmon, and striped bass) have increased over time. At other sites, results were variable. Nonetheless, the total riverine population of shad appears to be increasing, perhaps as a result of improved water quality and reduced exploitation as well as the installation of the fish passageways.

The present success of the Connecticut River program demonstrates that large numbers of American shad and blueback herring can be restored to areas upstream of hydro dams. Restoration of existing stocks of shad has largely resulted from the installation of upstream and modest downstream fish passage facilities. For Atlantic salmon, where no population was present, restoration appears promising, and a natural breeding population of salmon could eventually be restored to a river they have not inhabited for almost 200 years. Since 1974, when one adult returned, the number of returning adults to the Connecticut River has increased to a record of 529 in 1981. Eventual goals of the program are to produce about 200,000 wild Atlantic salmon smolts per year within the river basin and insure that 2,000 adult salmon in excess of spawning needs are available for an annual sports harvest (Minta et al., 1982).

The Connecticut Department of Environmental Protection (1985) reported that the goal of the Thames River restoration program was to provide and maintain a sport fishery for American shad and Atlantic salmon in the river basin and restore, enhance, and maintain spawning populations of anadromous fish species in all suitable habitats. Water quality has improved considerably since domestic and industrial pollution sources are under control. On the other hand, no fish passage facilities existed at any of the dams in the Thames River watershed in 1985 at the start of the project. A first step in restoration was to prioritize those dams that should be considered for fish passage. Populations of American shad, river herring and Atlantic salmon can be expected to increase because of the added production in areas made available by the implementation of fish passage at each succeeding barrier. It may be necessary to supplement natural spawning with hatchery-produced fry, parr, or smolts to meet recreational fishing demands. A final consideration will be the establishment of minimum flow requirements to provide for the needs of returning fish. The ultimate goal of the project is to restore 450,000 American shad and 8,000 Atlantic salmon to the system. The current level of success for this restoration project was not evaluated.

Stahlnecker et al. (1989) and Stahlnecker and Squires (1991) described a plan for the restoration of alewife, American shad, and Atlantic salmon to the Kennebec River. The restoration plan reflected conditions set forth in a cooperative agreement between the state of Maine fisheries agencies and the Kennebec Hydro Developers Group (KHDG). This agreement facilitates restoration by setting dates for fish passage and providing of funds by KHDC to fully implement an interim restoration program for 1986-1999.

The goal of the restoration plan is to restore alewives, salmon, and American shad to their historical habitat above the Edwards Dam in Augusta. The long term objectives are to achieve an annual production of 6.0 million alewives and 725,000 American shad above the dam in Augusta. For alewives, the strategy involves an interim trap and truck program fully funded by KHDG. This program initially involves stocking alewives at a rate of six adults per surface acre in ten lake systems representing about 50% of the alewife habitat historically available. The interaction between alewives, shad, and salmon will then be assessed to determine if restoration to the remaining lakes will occur.

For shad, restoration involves the passage of shad through a requested passage facility at the Edwards Dam and/or supplemented by trapping and trucking of adult shad from the lower Kennebec River or from out-of-basin for the interim period 1986-1998. After the interim period ends, fish passage will be provided at all mainstem dams and tributary dams as outlined in the Plan and Agreement.

The interim plan for Atlantic salmon calls for the passage of whatever Atlantic salmon become available at the Augusta dam into the upriver headpond and trapping at Augusta and transport to selected upriver areas. The KHDG Agreement provides for the attempted capture of Atlantic salmon below the Edwards Dam if no passage is available in order to accelerate restoration of this species in the Kennebec River.

The license for the dam at Augusta expires in 1993. In 1989 an experimental fish pump was installed at the dam but proved ineffective in capturing sufficient adult fish for stocking in upriver lake systems. The state of Maine is in favor of removal of this dam to restore the river segment above it as a spawning and nursery area for all anadromous fish species, including striped bass, rainbow smelt, shortnose sturgeon, and Atlantic sturgeon, which do not use conventional fish passage facilities. It appears that it will be necessary for the near future to continue to obtain broodstock from other sources.

The results of the restoration efforts follows. For alewife, the stocking goal of six fish per acre was achieved for a single pond and ranged from 47-97% of the stocking goal in the other five lakes. Juvenile alewives were collected in all ponds stocked with adults in 1987 and five ponds in 1988. The downstream emigration of juveniles was subsequently monitored. Passage or discharge was typically available at some sites, while occasionally quick interim action by developers was required to provide downstream passage. In other cases no opportunity for passage occurred.

American shad were obtained from the Narraguagus River to supplement the shad brood stock obtained from the tidal portion of the Kennebec drainage. If adequate interim trapping and passage facilities were installed at Edwards, there could be a significant number of American shad passed upriver into the impoundment. This component of local stock would have a significant impact on restoration efforts, since large numbers of shad brood stock are so difficult to obtain in Maine. The minimum stocking goal of 500 shad was not achieved in 1987 and no juvenile shad were subsequently collected from the stocking area. In 1988, the brood stock from the Kennebec Rivers was supplemented with fish collected from the Connecticut River and 899 shad were transported by tank truck to the upper Kennebec. A single juvenile was collected in 1988 indicating that spawning had occurred. However, it was impossible to estimate how many juveniles were produced. One juvenile shad was captured in 1989 and none in the impoundment above Edwards mill in 1990.

The objective for shad was to pass 2,500+ adults a year at the Augusta dam. Since 1987, fish passage for shad at the dam has been non-existent or ineffective. Although shad have been obtained from other sources, as noted above, the numbers stocked have not approached the goal. Stahlnecker and Squires (1991) noted that unless new sources become available, the goal for American shad is currently to stock 1,000 adults annually.

Only a single salmon (a returning hatchery stray from another river) was collected from below the Augusta Dam in 1987 and stocked above the dam. Since only one adult was moved no natural reproduction occurred. In 1988, 17 salmon were trapped below the Augusta dam and moved above it. No record was made of juvenile production. The fish pump system at Edwards Mill did not capture any Atlantic salmon during 1990. Large schools of salmon were visible swimming in the area of the pump intake, but no trapping occurred. Dozens were often sighted at one time. Clearly, large numbers of salmon could have been moved above the dam if workable trapping/sorting/passage equipment were in place.

3.3.2.5.7.2 Discharge Water

Smallowitz (1989) and Williams and Tuttle (1992) describe the reestablishment of anadromous fish populations in the Columbia River Basin. The Columbia River currently supports only 15% of the estimated 10-16 million annual run of salmon and trout, which was the average a century ago. An estimated 75% of the losses are a result of hydro development. The Northwest Power Act was established a decade ago to improve fish stocks in the basin. (Other relevant legislation is reviewed by Williams and Tuttle, 1992.) The original goal to boost fish runs by 5-11 million fish was not met and was replaced by an interim goal of doubling the existing fish population. A major goal of the restoration plan is to build mechanical devices to get fish past the dams (upstream and downstream) without harm. Since installation of such projects may take up to ten years, a temporary two-part plan is in effect to use water to get fish through the basin. A flow program, called the water budget, sets aside water that can be released to help young fish get through slow moving reservoirs before time disrupts their natural migratory cycles. A second agreement is in place to discharge water over the dams, safely washing young fish to the sea rather than forcing them to travel through turbines. Currently, this plan is costing $100 million a year, most in the form of money not earned in power revenues due to the water diversion.

Progress was slow for a variety of reasons. Modifications to existing dams will cost $250 million and extensive long-term testing is needed to determine the effectiveness of planned modifications. In addition, funds were lacking (i.e., the total cost of restoration over 20 years was estimated as high as $ 1 billion) and the problem of improving the fish situation without affecting agriculture is difficult to solve. Nonetheless, fish runs have improved in some areas although it is difficult to quantify the numbers and harder still to attribute the reasons for the successes. Increases could be a result of natural cyclical variation or other unrelated factors such as fishing treaties limiting overfishing.

3.3.2.6 Acquire and Protect Habitat

One means of encouraging recovery of resources and services injured by discharges is to provide additional protection to important habitat. An initial step is to determine which areas are the most important to fish. Some suggested methods for implementing protection measures include:

- Purchase of land. This should be based on a prioritized list of private lands in discharge area that are scheduled for development within five years and information that indicates potential for expanding anadromous fish resources in candidate areas;

- Purchase of conservation easements to landowner agreements. Conservation easements involve purchase of certain rights to use land, e.g., standing timber, without the purchase of the land itself. Development such as clear cut logging is a potential threat to salmon spawning environment. Loss of spawning habitat will further impede recovery or inflict additional injury; and

- Changes in future land management actions. Example: Assess habitat value of streams that are scheduled for land use alteration in near future.

3.3.2.7 Fish Restoration and Recovery: Summary and Conclusions

3.3.2.7.1 Summary of Effectiveness and Success of Actions

For offshore marine fish species, the most appropriate restoration technique is to permit natural recovery to occur. Offshore species appear able to avoid oiled areas following a discharge and fish kills among them were not recorded. While there can be a heavy loss of pelagic eggs and fish larvae if present at the time of a discharge, in most cases this mortality has had no detectable impact on fish stocks or catch.

Historically, the most widely used technique for enhancing fisheries was to use a spectrum of regulations to control harvest. However, most authors have concluded that these restrictions are limited in power to increase the resources available for harvest or to affect the temporal or spatial distribution of these resources (Buckley, 1989). However, these standard techniques are frequently successful when used in combination with other restoration and enhancement approaches such as habitat improvement, pollution abatement, and/or stocking.

The successful enhancement of anadromous and freshwater fish species is historically linked to artificial production in hatcheries. In the early era of fisheries management, hatchery propagation and restocking were perceived as the preferred technique for restoring fish runs depleted by over-fishing, pollution or stream degradation. Unfortunately this approach may result in the demise of many populations of wild genomes. The desirable long-term goal for restoration is to maintain the existing wild stocks and preserve genetic variability.

Restoration of anadromous and freshwater fish populations is currently perceived of as a three step process:

- A program of watershed protection, including:
    - ♦ Water quality control;
    - ♦ Ccontrol of erosion;
    - ♦ Restoration and maintenance of natural flow regimes; and
    - ♦ Revegetation and second-growth management;
- Stabilization of stream channel and instream habitats to restore habitat carrying capacity; and
- Management of fish resources, including:
    - ♦ Limitation of harvest;
    - ♦ Construction of spawning and egg incubation channels to restore or enhance reproductive capability of streams;
    - ♦ Establishment of side channels to increasing spawning habitat; and
    - ♦ Predator control.

Marine fish stocking programs were recently reviewed by MacCall (1989) who concluded that marine hatcheries have a long history of expensive operation with no demonstrable positive effect on the resource. The few cases where marine fish hatcheries seem to have produced recoverable fish were associated with estuarine rather than open-ocean fisheries. A major problem with the approach, in general, is that it remains extremely difficult to detect the survival rate of hatchery produced fish. Modern techniques of genetic marking and fingerprinting provide new tools for determining hatchery success but are currently extremely expensive to implement.

Marine fish hatcheries may be a functional and productive enhancement option only for the few species that have accessible spawning aggregations, culturable embryonic and larval development, and adaptable juvenile rearing stages. In addition, the demand for these species must justify continual large amounts of capital and operational funding (Buckley, 1989). Thus far these conditions have only been met by a single estuarine species in the United States, red drum. Interest in enhancement of marine fish stocks by release of hatchery-produced fish is at an all time high. The successful restoration of marine fish stocks by this approach will be evaluated (Willis and Roberts, 1991) in the future, but the overall usefulness of the technique remains in question.

Habitat loss and/or degradation is one of the principal reasons for the decline in a number of living marine resources. An active program of habitat restoration and creation involves more than just cultivating vegetation, breaching dikes, transplanting corals or nourishing beaches. Even where there are documented successes in habitat restoration or creation, there are problems that need to be addressed regarding the ability to restore functional attributes of habitats to the level of natural habitats (Thayer, 1992). Research may eventually demonstrate that the design criteria for projects need only be improved to approach the functional levels of natural habitats. On the other hand research may show that we cannot emulate nature as easily as has been assumed. The application of restoration techniques must acknowledge the need for research and/or pilot studies and scientific monitoring to determine their success (Thayer, 1991). Some of the ongoing fish habitat restoration projects described in the report (i.e., creation of intertidal and wetland habitat) may eventually reveal the viability of these techniques for enhancing fish populations.

The actual creation of new habitat may prove more difficult than restoring injured habitat. However, even restoring injured habitat is not always simple and/or successful. For example, effects of acid rain on aquatic communities may be temporarily mitigated by chemical neutralization techniques (for review, see Fraser and Britt, 1982). However, Zurbach (1984) found that many mitigation techniques currently used are inefficient, short-lived, and expensive. Mitigating stream acidification presents special difficulties because water must be neutralized on a continuous basis, at least seasonally. Pumping alkaline groundwater appears to provide a relatively inexpensive alternative to limestone addition and should be considered when and where possible. Despite problems, neutralization of acidity in lakes and streams remains the major approach to preserving or restoring aquatic life in poorly buffered waters (Schreiber and Britt, 1987).

Even the restoration of water quality may only lead to partial recovery of fish communities, at least over short time scales. Full recovery may eventually occur but is dependent on sufficient time for food organisms to become reestablished and time for native species to reinvade from outside the area of impact.

Altering marine habitats to increase fishery productivity is well within current technological capabilities (Buckley, 1989). The two most common methods of marine habitat alterations, artificial reefs and fish aggregating devices (FADs), enhance marine fisheries by increasing the amount of marine resources available for harvest, controlling the temporal and spatial distribution of these resources, and supplementing natural production and recruitment of reef-related species. Regrettably, most current applications of and designs for artificial reefs are flawed, primarily due to the prevalent use of materials of opportunity to construct the reefs. The first criterion must be the careful evaluation of realistic and justified fishery enhancement objectives that are the bases for habitat alteration. These objectives must address the species to be enhanced, fisheries that will benefit, and potential for adverse impact. In addition, appropriate siting and design criteria must be applied. The physical and spatial designs of artificial reefs must consider habitat configurations that allow replication of natural reef systems, especially for the recruitment, survival, and growth factors that control production. If properly applied, these approaches (i.e., artificial reefs and FADs) may be successfully used to restore fish populations lost from reef or offshore bank habitats.

Construction of artificial stream structures (e.g., log weirs or gabions), in contrast, has not proven to be a successful technique for restoring fish populations in streams altered by logging. In many cases, structural modifications were either ineffective or even injurious. To function successfully, artificial structures must meet carefully-defined objectives specific to target species, life history stage and prevailing physical conditions, and design of such structures must be closely tailored to eomorphic and hydraulic conditions. To meet both biological and economic objectives, the gabions and log weirs must remain intact at the installation site for their projected life span (i.e., 20-25 years). In many cases, artificial structures not only failed within this time range but also caused inadvertent adverse physical effects (Frissell and Nawa, 1992). The wide range of failed structures indicates that simple changes in structural design or materials are unlikely to overcome the problem of high failure rates.

Frissell and Nawa (1992) concluded that the restoration of fish populations in the most productive fourth-order and larger alluvial valley streams will require the re-establishment of natural watershed and riparian processes over the long term. They recommend that restoration programs for these habitats follow a hierarchical strategy that emphasizes:

- Prevention of slope erosion, channelization and inappropriate floodplain development, especially in previously unimpacted habitat refugia;

- Rehabilitation of failing roads, active landslides, and other sediment sources (i.e., logged slopes); and

- Reforestation of floodplains and unstable slopes.

Several fish restoration techniques that have proven successful in a variety of habitats include: construction of impoundments, spawning channels, and dredge spoil islands. Knox et al. (1979) summarized restoration and/or enhancement techniques associated with channelization as:

- Installation of sediment traps to prevent sediment from leaving a construction site;

- Construction on only one side of a stream channel;

- Removal of waterway obstructions with hand tools and small equipment to minimize impacts;

- Construction of continuous pool-riffle fish habitat in bedrock;
- Installation of fish pools with deflectors and constructed riffles in earth sections;
- Woody and herbaceous plantings;
- Maintenance of shade over water;
- Wetland acquisition; and
- Use of fencing and vegetation markers.

Installation of fish passageways is also a proven technique for restoring anadromous fish resources. The major fish passageway improvements are ladders, hoists, transport fumes, trap and truck, step pool structures, and discharge water. Again this technique works best when used in combination with other approaches such as water quality improvement and reduced exploitation. The technique is also most promising when existing populations of the target species are already present. However, it is potentially successful even where historical populations are eliminated when combined with a brood stocking and hatchery rearing program.

Harvest refugia may be extremely effective fishery enhancement tools. A comparison of areas that are protected from exploitation either by regulation or inaccessibility shows that resident fish species are more abundant and reach a larger size in protected areas (Cowen, 1983; Cole et al., 1990). Control over spearfishing on heavily fished reefs can result in dramatic recovery of targeted species of reef fish. At Hanauma Bay, Hawaii, where spearfishing is banned, large schools of reef fish occur. Outside the sanctuary, large reef fish are rarely sighted. Reef fish have also responded to control over spearfishing in Florida. Evidence from coral reefs in the Philippines (Alcala,1988), a temperate ecosystem in New Zealand (Jeff, 1988), and marine reserves in Florida provide additional encouragement for this approach. Selection of refuge sites is critical and should be based on protecting ecologically discrete zones which can produce larval and juvenile recruits for harvest in adjacent zones. Additionally, empirical evidence should be gathered to assure that the most valuable resources are those afforded the greatest degree of protection.

In general, restoration of fish populations is a young and still unproved science. In many instances, it is still not possible to evaluate the success of a given restoration technique because pilot projects are incomplete, there are no controls for quantitative comparison, and no additional restoration techniques were attempted for comparative purposes. The best approach to restoring fish populations appears to be a systems approach (i.e., one that clearly defines objectives, specifications and quantitative criteria for evaluation of success).

3.3.2.7.2 Evaluation of Actions

The following is a checklist which may be used as criteria for decision making with respect to fish resource restoration (Exxon Valdez Oil Spill Trustees, 1992a). Sections 5 and 6 contain more discussion on this topic.

- Evaluate injury to spawning habitats and fish stocks:

- Analyze the ability of resource to recover naturally;

- Demonstrate effectiveness of the restoration technique;

- Estimate increase in fish production resulting from each proposed restoration technique;

- Estimate the importance of increase in fish production for various user groups, i.e., sport, commercial and subsistence groups;

- Estimate potential for the proposed project to maintain genetic characteristics of the affected population;

- Assess level of genetic damage within stock. For example, there is concern that genetic damage to salmon eggs and fry during the *Exxon Valdez* oil discharge could reduce productivity and fitness;

- Require future project maintenance;

- Analyze ability to document success of project;

- Consider compatibility of project with established land/water uses in area; and

- Consider compatibility with regional management plans.

3.3.3 Reptiles

No evaluation was made on restoration of reptile species other than sea turtles. While little literature exists on restoration of most freshwater reptiles, crocodilian species in the U.S. have been depleted in the past and restoration programs exist.

Sea turtles are highly migratory and inhabit the world's oceans. Under the Endangered Species Act of 1973, all marine turtles are listed as endangered or threatened, e.g., loggerhead, green, olive ridley, Kemp's ridley, leatherback, and hawksbill. The NMFS has authority to protect and conserve marine turtles in the seas and the U.S. Fish and Wildlife Service maintains authority while turtles are on land. The Kemp's ridley, hawksbill, and leatherback turtles are listed as endangered throughout their ranges. The loggerhead and olive ridley turtles are listed as threatened throughout their U.S ranges, as is the green turtle, except the Florida nesting population, which is listed as endangered.

Historical data on sea turtle numbers are limited. In addition, the length of time over which data were collected is short when compared with the long life and low reproductive rate of all turtle species. It is difficult to assess the long-term status of sea turtles due to the limited data.

Sea turtles are fully protected in U.S. waters, but their habitats continue to be degraded. Coastal development is reducing nesting, nursery, and foraging habitats. Experimental and field results reported by Vargo et al. (1986) indicate that marine turtles would be at substantial risk if they encountered an oil discharge or large amounts of tar in the environment. Physiological experiments indicate that the respiration, skin, some aspects of blood chemistry and composition, and salt gland function are significantly affected (Vargo et al., 1986). Discharges in the vicinity of nesting beaches are of special concern and could place nesting adults, incubating egg clutches (Fritts and McGhee, 1989), and hatchlings at significant risk. Exploration and oil development on live bottom areas may disrupt foraging grounds. The U.S. Coast Guard has contingency plans for the containment, recovery, and minimization of injury from discharges of oil and hazardous substances. One source of direct mortality for turtles is the removal of oil rigs in the Gulf of Mexico (Murphy et al., 1987). Generally unused rigs are blown up below the surface of the seafloor and cause turtle strandings on Texas beaches.

Exploratory oil and gas drilling may affect sea turtles by attracting them to lighted platforms where they may be susceptible to increased predation, by disrupting feeding habitats when disposing of drilling mud and sediments, and by discharging oil that may contaminate turtles and cause injury to eyes, affect respiration, and cause abnormal behavior.

Sea turtles have been adversely affected by petroleum and its tar residue (Fritts and McGhee, 1982). Turtles are non-selective feeders who unknowingly ingest tar. The immediate effect of ingesting tar appears to be mechanical in that it seals the mouth shut and may clog the nostrils. Additionally, the crude oil phase may have a toxic effect. Most petroleum-impacted turtles have been found on beaches. Individual turtles, if recovered soon enough, may be treated, but it is unknown if those impacted by liquid oil can be saved.

To aid an affected turtle:

- Gently scrap off excess tar by removing residual tar or oil from the body and mouth using vegetable oil, mineral oil or waterless hand cleaner;

- Use a cotton tipped swab to clean the mouth, taking care to clear the nostrils;

- Rinse with a mild etergent, followed by a clean water rinse;

- If tar appears to have been ingested, administer a small dose (1-2 ml) of mineral oil; and

- Keep the turtle in an aquarium following cleaning until fully recovered as indicated by active feeding and swimming prior to release.

In addition to floating tar balls, plastics, if eaten, can harm or kill sea turtles. The magnitude of the above problems is not fully known, but they occur worldwide and international cooperation for marine turtle protection and recovery is needed.

In the Pacific, there are concerns about sea turtle deaths in the high-seas driftnet fisheries. Turtles are also killed when accidentally caught in other fisheries. As many as 10,000 sea turtles may be taken annually in shrimp trawls. Turtle excluder devices (TED's) have been developed and, when attached to shrimp trawls, enhance turtle safety by releasing them. TED's reduce the turtle kill by shrimp trawls by 97% and their use is mandated for certain shrimp fishing areas. Studies indicate that the use of TED's has reduced shrimp catches only about 5-15%. Shrimpers are concerned about reduced income owing to lower shrimp catches.

Five factors have resulted in declining turtle stocks:

- Destruction or modification of habitat as a result of pollutants from industrial and residential development, exploratory oil and gas drilling, disposal of garbage at sea, dredge and fill, and power boats;
- Overuse for commercial, scientific, or educational purposes;
- Inadequate regulatory mechanisms;
- Disease and/or predation; and
- Other natural or man-made factors such as incidental catch.

Restoration documents exist for each sea turtle species. Although restoration plans are species-specific, six major actions are generally needed to restore sea turtle populations:

- Provide long-term protection to important nesting beaches;
- Insure 50% hatch rate (at least) on major nesting beaches;
- Implement lighting plans or ordinances on all major nesting beaches within each state;
- Determine distribution and seasonal movements for all life stages in marine environment;
- Minimize mortality from commercial fisheries; and
- Reduce threat from pollution.

The NMFS (1990) has outlined a recovery plan as follows:

I. Protect and manage habitats.

- A. Protect and manage nesting habitat.
  - a. Ensure beach nourishment projects are compatible with maintaining good quality nesting habitat.
    - 1. Implement and evaluate tilling as a means of softening compacted beaches.
    - 2. Evaluate the relationship of sand characteristics (including aragonite and hatch success, hatchling sex ratios, and nesting behavior.
    - 3. Reestablish dunes and native vegetation.
    - 4. Evaluate sand transfer systems as alternative to beach nourishment.
  - b. Prevent degradation of nesting habitat from seawalls, revetments, sand bags, sand fences, or other erosion control measures.
    - 1. Evaluate current laws on beach armoring and strengthen if necessary.
    - 2. Ensure laws regulating coastal construction and beach armoring are enforced.
    - 3. Ensure failed erosion control structures are removed.
    - 4. Develop standard requirements for sand fence construction.
  - c. Acquire or otherwise ensure the long term protection of key nesting beaches.
    - 1. Acquire or protect all undeveloped beaches which provide important habitat for maintaining the historic nesting distribution.
    - 2. Evaluate the status of the important nesting beaches.
  - d. Remove exotic vegetation and prevent spread to nesting beaches
  - e. Evaluate and implement measures to enhance important nesting threatened habitat by erosion or tidal inundation.
- B. Protect marine habitat.
  - a. Identify important habitat.
  - b. Prevent degradation and improve water quality of important turtle habitat.
  - c. Prevent destruction of habitat from fishing gears and vessel anchoring.
  - d. Prevent destruction of marine habitat from oil and gas activities.
  - e. Prevent destruction of habitat from dredging activities.
  - f. Restore important foraging habitats.

II. Protect and manage population.

A. Protect and manage populations on nesting beaches.

a. Monitor trends in nesting activity by means of standardized surveys.

b. Protect nests from predators via

1. use of wire enclosures;
2. chemical repellants; and
3. aversion conditioning of predators.

c. Evaluate nest success and implement appropriate nest protection measures.

d. Determine influence of factors such as tidal inundation and foot traffic on hatching success.

e. Reduce effects of artificial lighting on hatchlings and nesting females.

1. Determine hatchling orientation mechanisms in the marine environment and assess dispersal patterns from natural (dark) beaches and beaches with high levels of artificial lighting.
2. Implement and enforce lighting ordinances.
3. Evaluate extent of hatchling disorientation on all important regional nesting beaches.
4. Evaluate need for federal lighting regulations.
5. Develop lighting plans at Port Canaveral, Kennedy Space Center, Canaveral Air Force Station, and Patrick Air Force Base, FL.
6. Prosecute individuals or entities responsible for hatchling disorientation under the Endangered Species Act or appropriate state laws.

f. Eliminate vehicular traffic during nesting and hatching season.

g. Ensure beach nourishment and coastal construction activities are planned to avoid disruption of nesting and hatching activities.

h. Ensure law enforcement activities eliminate poaching and harassment.

i. Determine natural hatchling sex ratios.

j. Define geographical boundaries of breeding aggregations.

k. Continue evaluation of hatcheries and head starting programs.

B. Protect and manage populations in marine environment.

   a. Determine distribution, abundance, and status in the marine environment.

      1. Determine seasonal distribution, abundance, population characteristics, and status in bays, sounds, and other important nearshore habitats.
      2. Determine adult navigation mechanisms, migratory pathways, distribution and movements between nesting seasons.
      3. Determine present or potential threats to turtles along migratory routes and on foraging grounds.
      4. Determine breeding population origins for U.S. juvenile/subadult populations.
      5. Determine growth rates, age of sexual maturity and survivorship rates of hatchlings, juveniles and adults.

   b. Monitor and reduce mortality from commercial and recreational fisheries.

      1. Implement and enforce TED regulations in all U.S. waters at all times.
      2. Provide technology transfer for installation and use of TEDs.
      3. Maintain the sea turtle stranding and salvage network.
      4. Continue nesting population studies.
      5. Identify and monitor other fisheries that may be causing significant mortality.
      6. Promulgate regulations to reduce fishery related mortalities.

   c. Monitor and reduce mortality from dredging activities.

      1. Monitor turtle mortality on dredges.
      2. Evaluate modifications of dredge dragheads or devices to reduce turtle captures, and incorporate effective modifications or devices into future dredging operations.
      3. Determine seasonality and abundance of sea turtles at dredging localities, and insure that dredging is restricted to time periods with the least potential for turtle mortality.

   d. Monitor and prevent adverse impacts from oil and gas activities.

      1. Determine the effects of oil and oil dispersants on all life stages.
      2. Ensure that impacts to sea turtles are adequately addressed during planning of oil and gas development.
      3. Determine sea turtle distribution and seasonal use of marine habitats associated with oil and gas development areas.

e. Reduce impacts from entanglement and ingestion of persistent marine debris.
   1. Evaluate the extent of entanglement and ingestion of persistent marine debris.
   2. Evaluate the effects of ingestion of persistent marine debris on health and viability of sea turtles.
   3. Determine and implement appropriate measures to reduce or eliminate persistent marine debris in the marine environment.

f. Maintain law enforcement efforts to reduce poaching in U.S. waters.

g. Centralize administration and coordination of tagging programs.
   1. Centralize tag series records.
   2. Centralize turtle tagging records.

h. Ensure proper care of sea turtles in captivity.
   1. Develop standards for care and maintenance including diet, water quality, and tank size.
   2. Develop manual for treatment of disease and injuries.
   3. Establish catalog for all captive sea turtles to enhance utilization for research and education.
   4. Designate rehabilitation facilities.

i. Determine etiology of fibropapilliomatosis.

III. Information and education.

A. Provide slide programs and information leaflets on sea turtle conservation for general public.

B. Develop brochure on recommended lighting modifications or measures to reduce hatchling disorientation.

C. Develop public service announcements regarding the sea turtle artificial lighting conflict, and disturbance of nesting activities by public nighttime beach activities.

D. Ensure facilities permitted to hold and display captive sea turtles have appropriate informational displays.

E. Develop standard criteria and recommendations for sea turtle nesting interpretive walks.

F. Post information signs at public access points on important nesting beaches.

IV. International cooperation: Develop international agreements to ensure protection of life stages that occur in foreign waters.

Restoration of sea turtle populations injured by oil discharges should be performed in association with the above planning outlined by NMFS. Also, a Pacific basin sea turtle recovery plan is presently being prepared by Hubbs Sea World Research Institute for NOAA (Commerce Business Daily, Issue No. PSA-0890, July 19, 1993). In addition, NMFS plan is a model for restoration planning for other injured wildlife species.

3.3.3.1 Ridley

According to Marquez et al. (1989) the Kemp's ridley sea turtle is the most vulnerable of the sea turtle species for several reasons:

- It is unique in that its population is nearly completely confined to the Gulf of Mexico;
- It nests almost exclusively along a 60 km strip of sand beach on the northern gulf coast of Mexico;
- Its feeding behavior of seasonal wandering for food on the shrimp grounds brings it in contact with shrimp trawlers; and
- A part of the population, specifically the juveniles, migrates out of the Gulf of Mexico through the Florida Straits with the possibility of no return.

A controversial program to develop a second nesting colony at Padre Island, Texas, has continued since 1977 (Taubes, 1992). Unfortunately, none of the 18,000 headstarted turtles has been observed returning to a beach to nest, after 15 years of conservation effort. Since there is no means of marking turtles, there is no way to establish a control group of wild turtles to compare with them, and so no ability to establish the success or failure of the project. Headstarting should not be viewed as a viable restoration technique (and is not included in the recovery plan, see above) unless or until the long term survival of the turtles can be demonstrated and compared with the wild population.

3.3.3.2 Loggerheads

The loggerhead turtle is federally listed as threatened worldwide. Nesting in the U.S. occurs primarily along North Carolina (1.5%), South Carolina (8%), Georgia (1.5%), and Florida (89%) beaches and accounts for approximately one-third of the world population. Nesting trends are declining in Georgia and South Carolina, unknown in North Carolina, and appear stable in Florida (NMFS, 1990). Coastal development threatens nesting habitat and populations, while commercial fisheries and pollution pose significant threats in the marine environment. At some future date, sustainable losses may become predictable and manageable and the loggerhead may be removed from threatened status. Until then, known mortality factors must be mitigated until their individual and collective effects on population numbers can be measured. A series of potential indices of population numbers and vitality (numbers of nesting females, numbers of hatchlings per kilometer of nesting beach, numbers of subadult carcasses appearing on beaches, etc.) should be monitored. Taken collectively, these variables represent the best available approach to measuring loggerhead population vitality and response to management efforts (Hopkins and Richardson, 1984).

3.3.3.3 Green Turtles

Green turtles were listed as Threatened/Endangered under the Endangered Species Act in 1978. The species is also protected by state laws in coastal states. All of the Atlantic sea turtle populations are threatened except the Florida nesting population, which is considered endangered. Green turtles are considered the most palatable of all sea turtles. Nesting in some areas may have been eliminated by overuse of the resource from commercial harvest by fishermen. Records show drastic declines in the Florida catch during the late 1800's. Similar declines occurred in other areas. Current problems for the green turtle include coastal development of nesting beaches and other human activities, which are harmful to turtles of all sizes (Hopkins and Richardson, 1984). Factors considered particularly important to restoration of green turtle populations include management of natural beaches, regulation of petrochemical industry and bilge pumping, regulation of lights, foot traffic, ORV's, beach cleaning equipment, seawalls and beach nourishment projects, and enforcement of laws to prevent illegal harvest.

3.3.3.4 Hawksbill

The hawksbill occurs in southeastern U.S. waters, the Gulf of Mexico, the Caribbean and the Bahamas. There are a few nesting records for Florida and stray animals have been reported as far north as New England. Special emphasis is placed on recovery of hawksbills nesting on Caribbean Islands under U.S. jurisdiction (Hopkins and Richardson, 1984).

3.3.3.5 Leatherback

Leatherbacks frequent the entire Gulf of Mexico and the eastern coast of the U.S. as far as Canada. Nesting on the mainland is very rare and mainly confined to the Atlantic coast of Florida. Small but important nesting colonies occur in the Virgin Islands and Puerto Rico. The recovery plan addresses discrete nesting populations on Caribbean Islands under U.S. jurisdiction. Broad geographical areas that appear to contain significant numbers of non-breeding animals are also considered (Hopkins and Richardson, 1984). The deliberate taking of adults constituted a threat to the species, although egg collecting is the greatest threat. Other causes of mortality include long lines (Hildebrand, 1980) and ingestion of indigestible materials such as plastics (Mrosovky, 1982). One factor considered particularly important to leatherbacks restoration is protecting hatchlings during emergence. The outlined recovery plan is applicable to leatherbacks as well.

3.3.4 Birds

Seabirds and waterfowl are frequently injured following oil discharges. Death is caused by exposure after oil has destroyed the insulation that their feathers provide, poisoning from ingested oil, and physiological stress. Even small amounts of oil cause injury such as reduced hatchability of eggs or breeding failure. Review of the effects of oil on birds may be found in Seip et al. (1991), Jones et al. (1979), EVOS Trustees (1990c), EVOS Trustees (1992), White et al. (1979), Grave et al. (1977), and Szaro (1979).

3.3.4.1 Case Studies of Effects of Oil Discharges on Birds

Although the impact of oil discharges on bird populations has been recorded for many incidents, restoration has been planned only some incidents, most of them relatively recent (e.g., *Apex Houston*, *Presidente Rivera*, *BT Nautilus*, *Exxon Bayway*, *Amoco High Island*, *Nestucca*, and *Exxon Valdez*). Bird restoration planning following the *Exxon Valdez* oil discharge in 1989 is published and will be reviewed as an example.

<u>Murre Restoration Project</u>

Approximately 320 seabird colonies are present within the area affected by oil discharged by the *Exxon Valdez*. The colonies contained about one million breeding seabirds of which about three hundred thousand were breeding murres (U.S. Fish and Wildlife, Computer Archives 1986). Diving seabirds like murres are most impacted by discharges. The fact that these species are long-lived with low reproductive rates engenders concerns that recovery will be slow. An estimated three hundred thousand murres (including non-breeding and wintering birds in addition to breeding birds) were killed following the *Exxon Valdez* oil discharge.

A murre restoration project is being conducted by the U.S. Fish and Wildlife Service to monitor the recovery of breeding common and thick-billed murres in the Barren Islands and Puale Bay colonies on the Alaska Peninsula. The object of the study is to determine how fast murre colonies will recover and how recovery might be enhanced. For three years following the discharge there were reduced numbers of breeding murres, delayed reproductive chronology, lack of synchrony of egg laying, and low to no reproductive success (EVOS Trustees, 1992). Signs of recovery were seen in 1991. Monitoring is continuing.

Marbled Murrelet Restoration Study

Prince William Sound (PWS) was one of three major population centers in Alaska at the time of the *Exxon Valdez* discharge for the marbled murrelet, a small seabird that nests in old growth forests. An estimated 9570 were killed by the *Exxon Valdez* oil discharge (EVOS Trustees, 1992). Populations of murrelets have been declining substantially over the years and they are being considered for threatened or endangered status. Limited data is available on their breeding biology, but it is thought that their reproductive success is quite low. Their nesting habitat is also threatened by logging activities.

The *Exxon Valdez* restoration study will assess nesting habitat, behaviors, and vocalizations and activity patterns associated with nesting, so that criteria may be established for nesting habitat requirements. Censuses will also be conducted in PWS to locate areas used for nest sites. Protection of forested nesting habitat through acquisition is one approach being considered to aid species recovery, along with protection of shallow nearshore waters used for foraging during the breeding season.

Harlequin Duck Restoration and Monitoring

Harlequin ducks breed along mountain streams in coastal old growth forests. They have a relatively low reproductive rate because of a small brood size, second year sexual maturity, and low breeding frequency. They also have a high fidelity to breeding and wintering areas.

Harlequin ducks were heavily impacted by the *Exxon Valdez* oil discharge. There is evidence suggesting that sublethal effects of petroleum hydrocarbon contamination included reproductive failure (EVOS Trustees, 1992).

Protection and management of the population in the non-oiled areas of Prince William Sound has been considered to allow for later recolonization of impacted areas when oil levels in the intertidal areas are sufficiently low. In addition, protection and enhancement of undisturbed riparian corridors within timber sale areas may present loss or supply nesting habitat.

Harlequin ducks are among the least understood of waterfowl in North America. This restoration project will document nesting and brood-rearing habitat requirements. Such information would allow land acquisition and habitat enhancement choices to be made such that they facilitate recovery of populations.

Develop Harvest Guidelines to Aid Restoration of Harlequin Ducks

In 1993 the Alaska Department of Fish and Game will recommend harvest guidelines to facilitate restoration of harlequin ducks in Prince William Sound. 1991 Surveys have shown a population decline and near-total reproductive failure in oiled areas. Many ducks sampled remain in poor condition. Preliminary results of 1992 work suggest continued reproductive failure (EVOS Trustees, 1992c).

Protection of Bald Eagle Habitats

The U.S. Fish and Wildlife Service has been involved in protecting bald eagles and their habitat in Prince William Sound. Eagles feed in intertidal habitats and nest within 200 m of shore. The *Exxon Valdez* oil discharge killed an estimated 800-900 bald eagles and impacted their productivity (EVOS Trustees, 1992). This project involves a nest inventory in PWS along with identification of important feeding and seasonal concentration areas. Its goals are to identify and project threatened or important bald eagle habitats to ensure recovery of the PWS population and to maintain a healthy population. Information obtained from this study will assist in an overall habitat protection strategy for the discharge area which will help restore not only the bald eagle population but also other species dependent on timbered shoreline, old growth forest, and intertidal and riparian areas. It was provided data to justify lands for acquisition.

Monitor Marine Bird Populations in Prince William Sound

The U.S. Fish and Wildlife Service has conducted seabird population studies in PWS since the early 1970's. Of the species observed, cormorants, harlequin ducks, black oystercatcher, pigeon guillemot and northwestern crow populations declined after the discharge. The EVOS Trustees (1992) studies also have examined how reproduction and foraging ecology of these species have been affected and have examined hydrocarbon contamination in these species.

The goal of current work on this project is to obtain estimates of the summer and winter populations of marine birds to determine which populations are recovering. Such information is necessary to plan additional restoration actions.

Potential Impacts of Oiled Mussel Beds on Black Oystercatchers

The black oystercatcher is a large shorebird that lives rocky intertidal shores throughout the North Pacific. They nest on rocky points or inlets and feed on intertidal molluscs. In PWS they live on gravel shorelines and feed primarily on the mussel beds embedded in sand/gravel beaches. Studies have shown a decreased growth rate in chicks raised on oiled mussel beds even in 1992 EVOS Trustees, 1992).

The goal of work to be completed in 1993 is to determine if black oystercatchers breeding and feeding on shorelines are affected by oil persisting from the discharge, specifically if the oil is causing depressed growth rates. Such information can be used to identify habitats requiring additional treatment and to plan such restoration.

Pigeon Guillemot Colony Survey

The pigeon guillemot, a diving seabird found in PWS, feeds in nearshore waters and nests on rocky shores. The U.S. Fish and Wildlife Service has studied the population in PWS since 1970. This study aims to enhance recovery of the population by identifying important breeding areas for possible protection and additional cleanups (EVOS Trustees, 1992).

Guillemot nest sites are vulnerable to logging operations and shoreline development. Their foraging areas have been affected by logging, mining, intensive commercial fishing, barge and dredge operations, and recreational activities. Such areas near their colonies may be considered for protection and/or acquisition.

Enhancing the Productivity of Murres

Murres were the bird species most heavily impacted by the *Exxon Valdez* oil discharge in terms of numbers and percent of the population killed. Monitoring studies since the discharge have shown abnormal breeding behavior and low reproductive success. It has been shown that increased breeding success is seen with breeding in high-density concentrations and with laying eggs in synchrony with neighbors. Murres at colonies affected by the oil discharge have not yet resumed normal breeding cycles for reasons not yet understood. It is thought that the use of tape-recorded murre calls, placement of decoys, and dummy eggs might stimulate normal breeding behavior. A 1993 project conducted by the USFWS will evaluate the feasibility of using artificial means to stimulate normal breeding behavior. This will be measured by nesting chronology and success. Information obtained will be useful in the development of a management plan (EVOS Trustees, 1992c).

3.3.4.2 Direct Restoration Actions for Birds

Several techniques have been used to establish, re-establish, or augment wildlife populations. Many of these have concentrated on translocation or captive breeding to speed the rate of recovery of species after injury to a population and/or its habitat. Among the most well documented bird restoration efforts are reintroductions of birds of prey, particularly peregrine falcons and bald eagles. Since the mid-1970's, over 3000 peregrine falcons (Moser, 1990). and 300 bald eagles (Green, 1985) have been reintroduced in the United States. The work with these species has included: captive breeding, hacking, fostering, and recycling. Each has proven successful in increasing the numbers of the species of concern.

Captive Breeding

Two major captive breeding programs for bald eagles and peregrine falcons exist in the U.S. One at the US&FWS Patuxent Wildlife Research Center in Laural, Maryland, began in 1976. The second began as a cooperative effort between Cornell University and the Colorado Division of Wildlife. This breeding facility relocated to Boise, Idaho in 1984.

Eggs are incubated and chicks maintained in brooders and hand fed for a short period before being placed with captive foster parents. These young are released to wild foster parents or to hack sites. In a program in Oklahoma, eagle puppets are used for feeding to prevent any imprinting on humans prior to release.

Fostering

In "fostering" programs, young birds, typically a few weeks before fledging, are placed in nests of breeding pairs whose eggs have failed to hatch. These young are from captive breeding programs or the wild young of destroyed nests.

Hacking

"Hacking" involves the release of a captively-held raptor to the wild to sharpen its hunting skills with subsequent recapture. This is done in reintroduction programs where fledgings are released without adults to artificial nest sites. Food is provided surreptitiously until flying and hunting skills are developed.

In the absence of wild birds, hacking is valuable in the reintroduction of a species. Much higher rates of survival to fledging have been seen relative to simple release. Operation of a hacking facility is costly. However, as a population is reestablished it can be combined with fostering. As new breeding pairs become established after haking, a shift to fostering young into their nests can increase productivity and nesting density. This can be implemented until an optimal carrying capacity for a habitat is reached (Verser, 1990).

Recycling

When eggs are removed from a wild nest soon after laying the parent will often lay another clutch, or recycle. So far, nesting success has been poor after recycling. However, eggs removed can become part of captive breeding programs, with the aim of increasing overall reproductive success of the population.

3.3.4.3 Enhancement Actions for Birds

Following injury to a bird population, enhancement actions may also be considered. Relief from other stresses may enable a species to recover at a rate faster than without this assistance. The Restoration Planning Work Group for the *Exxon Valdez* oil discharge considered alternatives to restrict particular activities to reduce stress and protect habitats from future disturbances, as reviewed below (Versar, 1990).

Logging

Decreasing logging pressure could benefit a number of bird species by maintaining and protecting quality habitat. It might also reduce water-borne logging activities (storage, transportation, etc.) that affect intertidal and shoreline areas normally used by birds for feeding.

This could be accomplished by land acquisition or the purchase of logging and development rights. When not possible, creating logging-free buffers of an appropriate critical site along streams and the coastal perimeter would ensure nesting habitat for species such as bald eagles, falcons, great herons, owls, ducks, and mergansers (Versar, 1990).

Disturbance

Some kinds of disturbances during breeding periods can have a significant negative impact on bird colonies. In PWS, for example, disturbances include tourism, recreation, commercial fishing, air traffic, logging, human collection of eggs, and discharge cleanup activities (Versar, 1990). Enforcement of regulations to reduce disturbance, education of people to the effects of disturbances on marine breeding birds, and the designation of refuge areas is critical to reducing the negative effects of these activities. When disturbance can not be reduced in these ways, the trade-offs of more drastic measures that would reduce other services must be evaluated.

Commercial Fishing

Fishing can potentially stress seabird populations due to disturbance, competition for food, and direct mortality caused by lost gear. Many birds, such as murres, commorants, gulls, kittiwakes, guillemots, and eagles, rely on forage fish like herring. Thus, fishing of the species is in direct competition with these birds. Additionally, net-entanglement has been shown to be a significant source of mortality for seabirds. A greater understanding of the effects of disturbance and fishing competition along with the life history cycles of each species, must be sought so that appropriate management practices may be put in place.

Predation

Ground nesting bird species are particularly affected by introduced predators. Work on islands in the western Aleutians has shown a 400 percent increase in breeding birds in less than 10 years with fox removal. The problem is primarily predators that have been introduced, not native species. Strong management and removal of introduced predators would assist in bird population restoration. Effects of predator removal on other species and the ecosystem need to be evaluated before this actions is undertaken.

Chronic Oil Pollution

A variety of work has shown evidence of significant impact to birds from chronic pollution. Reductions in oil pollution by improved stormwater management and bilge cleaning practices might reduce some of this stress. Problems occur in harbors, near oil terminals, and in intertidal and subtidal forage habitats. Oil residues are passed through the food chain, impacting upper trophic level species.

Disease

Research to identify disease preventative methodology would be helpful in maintaining and improving productivity of birds.

Hunting and Egging

Local hunting and egg collections can cause a substantial stress to a population. Population status and dynamics must be understood, along with the magnitude of harvests, for a justifiable hunting plan to be developed. A reduction in egging pressure in areas where this practice is common can be a restoration option for many species.

Other Pollution and Stresses

The effects of erosion, runoff and pollution from mining can greatly injure habitat quality for seabirds. Such effects could be studied, remedied, and/or regulations enforced to restore affected habitat. For example, in the *Presidente Rivera* discharge (1989, Delaware River), trustees plan to stabilize and protect an existing bird rookery from on-going erosion caused by ship wakes (Helton, 1993).

3.3.4.4 Habitat Replacement and Enhancement for Bird Restoration

Habitat enhancement techniques include construction of nest boxes, platforms, and islands. This traditional approach is used widely by wildlife managers to increase local bird abundance and productivity (Shapiro and Associates, 1992).

The U.S. Army Corps of Engineers (USACOE) has developed island habitats on dredged material disposal islands throughout the U.S. and studied vegetation succession and wildlife use. Their objective has been to investigate, evaluate, and test methodologies for habitat creation and management on dredged material islands. An extensive amount of literature exists on this work (Buckley et al., 1978; Schreiber et al., 1978; Soots and Landin, 1978; Scharf, 1978; Coastal Zone Resources Division, USACOE, 1979; Landin, 1978). Most of this documentation is available at the USACOE library in Vicksburg, MS. The most significant wildlife aspect of these islands is their use by colonial nesting sea and wading birds such as gulls, terns, egrets, herons, ibises, and pelican's (Lewis, 1978).

As natural barrier islands and intertidal areas have been altered for man's use, these dredged material islands have provided replacement habitat for birds. Colonial seabirds and wading birds are known to have nested on dredge material islands since their first creation in Tampa Bay in 1930. In 1978, fifty percent of the colonial nesting sea and wading birds in Florida were nesting on dredge material. Many more species were using the islands for feeding and nesting. The same use has been observed on islands in the Great Lakes and in all marine coastal areas. A New Jersey study reported 52,205 pairs of nesting colonial gulls, gull-billed terns, common terns, snowy egrets, and glossy ibises on dredge spoil islands (Buckley, 1978).

There are many examples of wetlands and impoundment creation with the aim of attracting waterfowl and increasing their production. Several of these are reviewed in Sections 3.2.1 and 3.2.2. However, to be successful, these creation projects need to be carefully planned and executed, as described in the above sections. It should also be noted that a habitat creation project is also habitat destruction. That replacement of habitat needs to be considered to provide the highest net benefit to all habitats and dependant natural resources.

3.3.4.5 Monitoring and Management of Bird Populations

The most common, effective management practice for recovery of seabirds is protection from hunting, egging and disturbance. Secondly, management of the availability of prey, specifically fish, has been considered. The abundance and availability of food are critical to seabird population growth. These direct restoration actions have been shown to promote recovery (Nur and Ainley, 1992).

Monitoring the recovery of seabird populations is very important. In their review of recovery of marine bird populations, Nur and Ainley (1992) discuss in detail the parameters to be observed. They state it is common practice in seabirds to monitor the breeding population rather than the entire population, but feel that monitoring both parts of the population is of great value. Additionally, knowledge of the primary demographic parameters (fledging production, adult survival, juvenile survival, proposition of breeders among adults) is critical in effective monitoring and management. Two criteria of recovery are the return to historical population size or return to the population size that would have existed in the absence of the perturbation. The latter is more appropriate if a population is changing over time for reasons other than perturbation caused by the oil discharge.

3.3.4.6 Recovery Rates of Bird Populations

Different taxonic groups display characteristic (intrinsic) population growth rates when recovering from a population density below carrying capacity. Life history data on species impacted by a discharge should be analyzed to obtain this information. Seabirds vary from approximately 10% to 19% growth rate depending on numbers of eggs per clutch and survival rates. Rates of recovery also vary with time after a discharge incident or other injury, with growth rates higher immediately after injury and slowing down as carrying capacity is approached (Nur and Ainley, 1992).

Recovery of a species population after reduction by half would require seven to eight years at a growth rate of 10% per year, four years at a rate of 19%. These values would be changed by immigration or emigration of individuals. They also assume a habitat suitable of sustaining the population (nesting sites, food, etc.).

Monitoring should be carried out until a population has reached pre-incident numbers and condition, or that population size it would be if the discharge had not occurred (if the population size is changing due to other causes).

3.3.4.7 Bird Restoration and Recovery: Summary and Conclusions

For many seabird populations it is difficult to quantify life history parameters and the effects of environmental impacts. Restoration planning for bird populations should review all existing data on the species of concern and include input from experts. Monitoring projects are currently underway in Prince William Sound that will provide valuable previously-missing information needed to plan effective restoration. This type of research may be needed to plan restoration efforts for wildlife in order for the restoration to be effective. Not enough is presently known about effectiveness of restoration actions to *a priori* recommend specific actions. However, if a critical, limiting life history stage can be identified for a species, enhancement of those needs has been proven successful. For example, if nesting sites or success is limiting, providing new sites or protection can boost productivity and recovery rate. Providing feeding habitat has also proven successful. Reduction of hunting pressure is likely to help recovery.

3.3.5 Mammals

3.3.5.1 Marine Mammals

Most documentation of injury to mammals resulting from oil discharges focuses on marine mammals, both due to the higher frequency of large marine, as opposed to inland, discharges and to the special status afforded marine mammals in the U.S. by the Marine Protection Act of 1972 and the Endangered Species Act of 1973. Thus, review of possible restoration alternatives and actions focusses on marine rather then terrestrial mammals.

3.3.5.1.1 Harvest Alteration

Marine mammals are managed under the Marine Mammal Protection Act of 1972 and the Endangered Species Act of 1973. Thirty-six species range the U.S. Atlantic and Gulf of Mexico waters and forty-two species occur in U.S. Pacific waters. Populations of marine mammals have suffered large reductions, sometimes to near extinction, during the past two hundred years. Sources of mortality include commercial harvest, subsistence fisheries, incidental or deliberate killing, and epizootics (Stewart et al., 1992). In many cases, recent population recoveries of pinnipeds and cetaceans have been linked to the cessation of either commercial harvesting and/or the reduction of indiscriminate or incidental killing. For example, harbor seal populations have been increasing (5-22% per year) in most areas where commercial or subsistence harvesting is low or absent (Harvey et al., 1990; Heide-Jorgensen and Harkonen, 1988; Olesiuk et al., 1990a; Stewart et al., 1988; 1992). Northern elephant seals have been increasing at about 14% per year (Stewart, 1992) and killer whales, off British Columbia and Washington, have an annual rate of increase of 2.92% (Olesiuk et al., 1990b). Reilly and Barlow (1986) estimated that dolphins could approach a population growth rate of 9%, while baleen whales have demonstrated annual increases of 3 to 11.6% (Payne et al., 1990; Bannister, 1990; Zeh et al., 1991).

The current status of most species is poorly known, but some, like the right whale, Mid-Atlantic coastal bottlenose dolphin, harbor porpoise, Northern fur seal, Northern sea lion, harbor seal, and Stellar sea lion are under stresses that may affect their survival. In some cases chronic pollution is thought responsible for reproductive failures and depressed populations (Helle et al., 1976; Reijnders, 1978; Zakharov and Yablokov, 1990). Information on incidental take of marine mammals in commercial fisheries is still incomplete (substantial undocumented mortality is a possibility) and an assessment of the effects of fisheries and other human activities on the ecosystem is a critical long-term concern.

For some species, declining numbers are believed to be due to a combination of incidental kills in fisheries, illegal shooting, and changes in the numbers and/or quality of prey. Except for the northern spotted dolphin, the dolphin kill in the eastern tropical Pacific tuna fishery has declined drastically since the 1960's. Monitoring is essential to see if dolphin populations increase. The current accidental annual kill of northern spotted dolphin (36%) will have to decrease for the population to rebound.

The harbor porpoise kill in California's fisheries declined from 200-300/year in the mid-1980's to less than 100/year after gillnet fishing ceased. The harbor porpoise kill by the Makah Indian tribal setnet salmon fishery off Washington declined when fishing effort (for salmon) was reduced. The presence of abundant prey resources and good quality breeding habitat are probably the most important factors that allow sustained population growth when exploitation ceases (Stewart et al., 1992). Overall, long-term population data demonstrate the potential of pinnipeds and cetaceans to sustain high rates of growth (2-21% per year) following population reduction, even to very low abundance, so long as breeding and foraging habitats are not degraded (Stewart et al., 1992). For many species, far too little data exist to judge if stocks are recovering or what management actions are needed to enhance the stocks.

3.3.5.1.2 Habitat Protection and Reserves

Some human activities may be affecting the recovery of marine mammal species. For example, adult female humpback whales with calves have apparently been abandoning traditional nearshore calving and calf rearing habitat near Maui, Hawaii, owing to repeated human interference or contact (NOAA, 1991). Humpback whales in southeastern Alaska were reported to switch feeding grounds coincident to increased human disturbance for vessel traffic in Glacier Bay (Marine Mammal Commission, 1979). Hawaiian monk seals changed hauling and pupping sites in response to human disturbance (Gerrodette and Gilmartin, 1990).

Allen (1991) reported that although harbor seal numbers were increasing at various Californian coastal sites, the population in San Francisco Bay has remained at a low, relatively constant level of 400-500 animals. Within the bay, 94% of the shoreline habitat preferred by harbor seals has been altered or lost by filling and diking (Josselyn and Buchholz , 1984). Indirect evidence suggests that habitat loss, together with pollutants and disturbance has resulted in a less numerous harbor seal population within the bay area than 30-40 years ago (Paulbitski, 1972; Risebrough et al. 1979; Alcorn and Fancher, 1980).

Haul out areas provide breeding and resting sites for congregations of seals and protecting these areas is an important measure for preserving populations. Strawberry Spit, in San Francisco Bay, is one of only 12 known haul outs in the bay and provides evidence that development pressures directly affect habitat use. Risebrough et al. (1979) estimated that the number of seals using the spit during winter 1975-76 represented about one-third of the bay's harbor seal population. The number of seals has since dropped precipitously due to human disturbance. Other authors (Johnson, 1976; Calambokidis et al., 1978) reported reduced reproductive success and site abandonment as a result of human activities. To mitigate development effects on Strawberry Spit, the developer agreed to sever the spit from the mainland to create a seal refuge separated from the residential portion of an expanded development project, excavate a new haul out site, 1000 feet north of the existing site, to serve as an alternative haul out, more removed from residential areas, construct an earthen berm, fence and landscaping at the south end of the "island" to serve as a visual buffer, post signs on the northern end of the residential development identifying the island as a sensitive wildlife habitat, and restrict the rear property line of the residences bordering the navigational cut to a minimum of 425 feet from the south edge of the existing seal haul out beach, restrict dredging and construction activities from April to October when seals are absent from the area. These measures were designed to minimize the effects of existing and future disturbance to the seals. In addition to creating a more restricted island, the new channel would divert boat traffic away from the haul out area.

Unfortunately, the mitigation measures, and particularly the severance of Strawberry Spit from the mainland, were not completed in time to stop or reverse the rapid decline and eventual abandonment of the site. The authors also noted that there was evidence that a depleted food resource may have contributed with disturbance to cause desertion. Disturbance may have depressed seal usage of Strawberry Spit to a point such that when Pacific herring failed to spawn, seals readily abandoned the site.

There has been no evidence of re-establishment by the seals on the island to date and probably a few years will be required to determine if the mitigation measures have been effective.

Reynolds et al. (1991) noted that restoration of manatees requires information, on their distribution, abundance, and critical habitats. Using this information, seasonal or year-round regulatory zones can be created to protect the manatees directly, as well as their critical habitat. Based on year-round aerial surveys in Tampa Bay and intensive shore-based surveys at power plants in winter, Reynolds et al. (1991) developed site-specific management recommendations to protect manatees and manatee habitat. They recommended establishment of a 300-m wide, slow-speed shoreline buffer along the entire upper and lower bay along the shorelines of Pinellas, Hillsborough and Manatee Counties and including inshore waters of southwest Manatee County and Boca Ciega Bay, establishment of a 1500-m buffer zone in areas with dense seagrasses and heavy manatee usage (all creeks, rivers, bays and bayous connected to Tampa Bay), and establishment of site-specific protection measures where manatees frequent locations with critical resources, such as warm water in winter (near power plants), freshwater and abundant seagrass for food. Such site-specific measures should include seasonal (November 15 -March 31) manatee protection sites with idle-speed zones and no-entry zones, as well as slow speed zones with marked channels for boat traffic.

3.3.5.1.3 Restored Wetlands

Allen (1991), in the process of describing the abandonment of the Strawberry Spit haulout area for harbor seals, noted that seals had begun to use a haul out site at Muzzi Marsh in Corte Madera, an adjacent area. This discovery is significant since Muzzi Marsh is a 51 ha wetland restoration project, initiated in 1976 and completed in the early 1980's. The restoration project involved breaching dikes and planting cord grass. Seals haul out on a level platform of mud and pickleweed at the eastern edge of the marsh, including a small peninsula and adjacent cove. The site provides deepwater access to seals at high tides when tidal mudflats are flooded. The site is coincidentally isolated from hikers at this time since the site is flanked by two breached dikes that are flooded at high tide (>+2.5 feet above mean sea level). Seals began to appear at the site in 1985, most likely after discovering the relatively undisturbed site while on foraging trips, after the restoration project was completed. Seals may have selected the site because of several factors, low exposure to human disturbance, suitable physical characteristics including access to deep water and a sloping substrate, and closeness to a reliable food source. Allen noted that seals may have used Strawberry Spit for the same reasons. Mitigation measures at the spit are an attempt to restore the habitat to the original conditions. The Muzzi Marsh restoration project demonstrates that seals can benefit from measures designed to restore ecosystems. Allen (1991) concluded her report with a series of recommendations for management of marine mammal populations (general and specific to harbor seals):

- Determine what constitutes an optimum haul out site for seals so that the degradation of habitat can be clearly defined and creation of future haul out sites can be undertaken with these factors in mind;

- Fully study the effects of human activities on behavioral responses and reproductive success of harbor seals, and other coastal species of marine mammals;

- Clearly define what constitutes a disturbance so that proactive management guidelines can be developed;

- Establish guidelines regarding acceptable distances for human activities in the vicinity of marine mammal habitat; and

- Remain active in ongoing planning for development projects and dredging in the vicinity of marine mammal habitat. Many of these recommendations are appropriate for restoration of populations adversely affected by human activities, either currently or in the future.

3.3.5.1.4 Relocation

Concerns about extending the range of sea otters (within their historic range) prompted a translocation program to establish a colony on San Nicolos Island, 90 km west of Los Angeles. Since 1987, 138 otters captured along the mainland coast have been moved. Fourteen otters have remained around the island, plus three young that were born there. Many of the relocated otters eventually returned to the vicinity of their capture, raising questions about the effectiveness of the program. Success of relocation has been low and whether or not translocations should continue is being debated. In addition, fishermen fear the return of sea otters to southern California, where they could impact highly profitable shellfish and sea urchin fisheries (Thayer, 1992).

3.3.5.1.5 Effects of Oil Discharges on Marine Mammal Populations

No long-term population effects of oil pollution on pinnipeds have been documented (or rigorously examined for long enough periods to do so) (Stewart et al., 1992). Vulnerability of cetaceans to discharges is highest for species with small ranges (coastal, ice-dwelling, and/or riverine habitats), limited diets, poor behavioral flexibility, and small populations (Stewart et al., 1992). For pinnipeds, stressed or nursing animals and recently weaned pups are most vulnerable. Sea otters and other fur-bearing mammals are the most vulnerable species.

An estimated 3,500 to 5,500 sea otters were killed by the *Exxon Valdez* oil discharge. Post-discharge surveys showed measurable differences in populations and survival between oiled and unoiled areas in 1989, 1990, and 1991. Survey data have not established a significant recovery trend. Dead prime-age animals were still found on beaches in 1990 and 1991 suggesting continuing effects (Strand, 1993).

Stewart et al. (1992) noted that resident populations of harbor seals and killer whales may have been affected during the 1989 *Exxon Valdez* oil discharge in Prince William Sound by inhalation of volatile, short-chain hydrocarbons, ingestion of oil, immediate destruction of prey resources, and long-term food contamination. Substantial numbers of harbor seals became oiled and some were exposed to toxic aromatic hydrocarbons in areas near the discharge source (Stewart et al., 1992). An estimated 345 seals were killed. There was a greater decline in population indices in oiled areas compared to unoiled areas in Prince William Sound in 1989 and 1990. This population was declining prior to the discharge and no recovery was evident in 1992. Oil residues found in seal bile were five to six times higher in oiled areas compared with unoiled areas (Strand, 1993). Stewart et al. (1992) conclude that reducing and strictly regulating subsistence harvest would most likely be the most effective means of stimulating rapid population recovery for harbor seals in the Prince William Sound area.

Killer whale numbers have declined in the area of Prince William Sound since 1989 with 13 known (photo-identified) whales reported missing from a well-studied killer whale pod. Some experts believe that circumstantial evidence links the loss of the 13 whales to the oil discharge. Other experts think the deaths are unrelated to the oil discharge (Strand, 1993). Additional studies were conducted on the distribution and abundance of killer whales in Prince William Sound to determine the relationship of the discharge to changes in whale abundance (Stewart et al., 1992). The affected pod (AB) has grown by two individuals since 1990 (Strand, 1993). Recovery must be defined for killer whales since little pre-discharge data exist for comparison with post-discharge conditions. Stewart et al. (1992) suggest that one definition of killer whale recovery might be whether or not animals have regained the ability to maintain self-replicating or growing populations. Long-term studies of abundance coupled with an assessment of seasonal movements of animals in and out of the area and the magnitude of immigration and emigration are thus required.

3.3.5.1.6 Rehabilitation of Individual Animals

For marine wildlife in general, inhalation of hydrocarbon vapors, as well as fouling by oil following a discharge, pose a risk to individuals. Effects of oiling depend on whether oil coated the body surface, was ingested, or aromatic hydrocarbons were inhaled (Stewart et al., 1992). Sea otters, unlike many marine mammals, lack a subcutaneous fat layer and depend on air trapped under their fur for insulation (Davis et al., 1988). Contamination by oil eliminates the air layer, allows water to penetrate to the skin, and reduces insulation up to 70% (Williams et al., 1988). Because of their vulnerability, methods have been developed to clean and rehabilitate otters (as well as equally-vulnerable birds). Rehabilitation of individual animals is more typically performed as part of response, but might be considered as a restoration action.

Davis et al. (1988) developed a method to clean and rehabilitate otters that might become contaminated during an oil discharge. Otters were immobilized by injection and placed on a wire meshed trough. Otters were washed for 40 minutes with a solution of Dawn dishwashing detergent which was diluted (1:16 in water) to facilitate rinsing. Earlier studies (Williams et al., 1988) established that Dawn was the most effective agent in removing crude oil from sea otter fur. An equal period of rinsing was essential to remove residual detergent and to restore the water-repellent quality of the fur.

Williams et al. (1988) concluded that sea otters that have had 20% of their surface area oiled can be successfully cleaned and rehabilitated. Oil contamination increases thermal conductance and requires an increase in metabolic rate that may exceed the ability of wild otters to maintain core body temperature. An oiled animal must be captured and taken to a rehabilitation center within one to two days to insure the greatest chance of survival. Proper cleaning procedures and normal grooming by the otter restore the insulation of the fur and allow metabolism to return to normal levels. If the otter fails to groom, then the fur wets and thermal conductance remains high. Veterinary care is important to prevent the development of secondary infection such as pneumonia. At least one to two weeks should be allowed for restoration of fur and recovery from the stress of oiling and cleaning, provided no medical problems develop.

3.3.5.2 Terrestrial Mammals

3.3.5.2.1 Case Histories of Oil Discharge Effects on Terrestrial Mammals

No literature documenting or evaluating restoration of terrestrial mammal populations injured by an oil discharge has been documented, with the exception of a few species affected by the *Exxon Valdez* oil discharge.

Five species, brown bear, mink, black bear, sitka black-tailed deer, and river otters may have been exposed to oil from the *Exxon Valdez* through foraging in intertidal habitats. Some oil contamination was found in deer and a yearling brown bear, but injury to bears and deer could not be quantified. Injury to mink was considered possible, but was not quantified. Several river otter carcasses were recovered and evidence was obtained that additional animals were contaminated. Radio-tagged river otters showed home ranges in oiled areas twice that of unoiled areas and were of smaller size, suggesting dietary limitation. There is concern that otters will continue to be contaminated through mussels, a part of their diet (Exxon Valdez Oil Spill Trustees, 1992a,c).

The only direct restoration option for terrestrial mammals that was considered by the EVOS Restoration Planning Work Group was the translocation of river otters to augment populations within and outside the oil discharge area. However, this option was rejected on two grounds, sufficient source populations exist for natural recovery to occur and translocating river otters could result in introduction of disease (EVOS Trustees, 1992a). The concern of introduction of disease is an important consideration whenever translocation of wildlife is contemplated.

The EVOS restoration for terrestrial mammals is to include two alternatives, natural recovery with monitoring to determine the rate of recovery and whether further actions are necessary and acquisition and protection of habitats that will reduce or eliminate other perturbations on the populations. Harvest management has been considered but is not being pursued at this time (Versar, 1990; EVOS Trustees, 1992a).

3.3.5.2.2 Possible Restoration Alternatives and Actions for Terrestrial Mammals

Natural recovery is the most viable option if a population is not greatly injured by an oil discharge. For species that are exploited, management or elimination of harvest would enhance recovery. Where necessary, restocking might be a viable action, but there is little or no experience in this for most species and the same caveats true for other wildlife restocking efforts would be applicable for terrestrial mammals. Translocation is also possible, but introduction of disease must be controlled. Enhancement of habitat is likely viable. However, careful study of critical and limiting habitat requirements should be made in order to appropriately design enhancement actions for them to be effective. Likewise, protection of critical habitat to prevent future loss may be considered for restoration.

3.3.6 Monitoring the Recovery of a Species (Biological Natural Resource)

The issue of monitoring single species' recovery is beyond the scope of a narrow simplifying discussion. It will be largely dependent on the species, its habitat, and the chosen restoration action. The general guidelines for monitoring habitat recovery (Section 3.2.10) are relevant with some modification:

- Monitoring must be sufficiently long-term to ensure full recovery to a stable condition;

- Monitoring should sample all components of the environment related to the nature of the restoration action. All life stages of the species being restored must be quantified along with any environmental variables that may have been manipulated to effect the restoration;

- Appropriate control or reference information is needed to verify when restoration has occurred. In many cases, this may be data on the conditions predating the injury;

- The monitoring plan must be designed to produce statistically defensible results; and

- The plan should be sufficiently flexible to permit mid-course alterations if necessary.

Refer to the separate discussions of restoration actions for the various species for more specific information on.

4.1 Overview of Costs

This chapter presents information on the economic costs of identified restoration actions that were:

- Available from previous restoration activities after actual oil discharges. Based on the research conducted, it was found that available cost information from actual restoration activities in response to oil discharges was rather limited.

- Developed from additional sources on costs of potential restoration actions from non-oil situations. Because information on actual costs of restoration activities after actual oil discharges was limited, it was necessary to develop information on costs using data from non-oil situations.

The material presented in this section follows the outline for the material presented in Chapters 2 and 3. Potential restoration actions were identified previously (See Exhibit 2.2).

Potential restoration actions cover a diverse group of activities. As discussed in Chapters 2 and 3, actions include such activities as:

- Natural Recovery;

- Replanting in a number of different environments;

- Supplementary methods to remove residual oil contamination or mitigate further injury, such as cropping vegetation, constructing erosion control structures in saltmarshes, or opening of channels in mangrove swamps;

- Bioremediation to reduce the residual oil contamination;

- Activities specific to certain structured habitats such as reconstruction or reseeding of oyster reefs or coral transplants; and

- Activities for removing residual contamination from shorelines, such as flushing, sediment washing, sediment agitation, or incineration.

Wherever possible, cost information was extracted from a detailed review of the literature. However, certain costs were not available directly from published sources. In these instances, a considerable effort was devoted to developing cost estimates based on personal communication with knowledgeable experts. In some instances a significant degree of analysis was required in order to synthesize meaningful cost information. Cost information taken from historical sources was converted to mid-1992 dollars using relative price indices. (June 30, 1992 Producer Price Index.)

4.2 Economic Costs of Restoration Actions

4.2.1 Estuarine and Marine Wetlands

4.2.1.1 Saltmarsh

Saltmarsh restoration actions include:

- Natural Recovery;
- Replanting;
- Supplementary Erosion Control Structures;
- Sediment Removal/Replacement;
- Vegetation Cropping;
- New Saltmarsh Creation;
- Low Pressure Flushing; and
- Bioremediation.

Some actions will typically be performed in addition other Salltmarsh creation is included as an off-site replacement action. It is often coupled with replanting.

4.2.1.1.1 Oil Related Literature

Actual costing information reported in the literature for specific restoration actions after an actual oil discharge were identified in two cases. The first was reported in Krebs and Tanner (1981). This involved a discharge of No. 6 fuel oil in a smooth cordgrass (*Spartina alterniflora*) saltmarsh near the mouth of the Potomac River where it enters Chesapeake Bay. The restoration action consisted of sediment removal, disposal of the sediment, and backfilling with new material coupled with replanting of smooth cordgrass (*Spartina alterniflora*). It should be noted that many observers recommend against sediment removal except in extreme situations (Getter et al., 1984; Chapter 3).

The other case involved the *Amoco Cadiz* discharge in France as reported in Seneca and Broome (1982), Getter et al. (1984), and Broome et al. (1988). The restoration action consisted of replanting of several species of local saltmarsh vegetation.   Actual costs were not reported, but level-of-effort data were reported on the labor requirements for digging, separating, and transplanting.

4.2.1.1.2 Non-oil Related Literature

A number of literature sources include cost data that may be applicable to restoration efforts following an oil discharge. Much of this cost data is derived from wetland restoration work involving highway and other construction project mitigation efforts as well as work involving wetland creation on dredge spoil sites by the U.S. Army Corps of Engineers (USACOE). Key cost information on replanting and erosion control structures was extracted from USACOE (1978), Garbisch (1978), Broome et al. (1988), and Jerome (1979). Historical cost data on saltmarsh wetland creation projects was summarized from publications including Josselyn (1982), USACOE (1978), Josselyn et al. (1991), and Purcell and Johnson (1991).

4.2.1.1.3 Costs of Restoration Actions

Costs of saltmarsh restoration actions are discussed in the following subsections.

4.2.1.1.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.1.1.3.2 Replanting

It is noted in U.S. Army Corps of Engineers (1978) that marsh replanting costs can be expected to be extremely site-specific and will reflect such factors as:

- Logistics;

- Labor-hour costs and efficiency;

- Planting design (including the density of transplants); and

- Texture of the substrate.

Exhibit 4.1 presents labor requirements for planting in saltmarsh habitats. A key factor in the labor requirements is the planting density. The level of effort involved tends to be proportional to the number of plants. Planting labor requirements are given for spacings of 0.5, 0.6, and 1.0 meters. These correspond to 40,000, 28,000, and 10,000 plants per hectare, respectively. USACOE (1978) suggests that spacings of 1.0 to 1.5 meters will result in cover in one to two years in most situations. Closer spacing will be desired if faster plant cover is required, or if erosion presents potential problems. Level-of-effort data were typically reported for one of the spacings presented. These reported data were adjusted to the other spacings by extrapolating based on the number of plants.

There are many other projects reported in the literature for which brief cost summary information is provided. Most of these typically have much lower costs. It appears that most of these cases involve much less comprehensive restoration efforts and typically are associated with wetland mitigation projects.

The two main methods of revegetation include seeding and transplanting of sprigs, plugs, or potted plants. Seed or transplant propagules may be purchased from a commercial nursery or obtained locally from a donor location.

Individual task elements for marsh re-vegetation efforts include:

- Acquiring transplant propagules (either through purchase or digging and separating) if transplanting is the chosen action;

- Actual planting of the propagules;

- Purchasing or gathering of seed, if that is the chosen action;

- Seeding;

- Fertilizing (usually); and

- Follow-up effort including monitoring and selective replanting.

As seen in Exhibit 4.1, seeding is much more economical although, as noted in Chapter 3 is less effective. In general, reported labor requirements for the manual planting of sprigs, plugs, or potted plants range from 50 to 250 hours (for 1.0 meter spacing of plants). These figures quadruple as spacing is reduced to 0.5 meters. Manual digging and separating of plants reportedly requires from 50 to 133 hours. Mechanized digging (using an adaptation of a small agricultural tractor), separating, and planting requires half the time (Broome et al., 1988)

The low figures are from Broome et al. (1988) and represent the experience of skilled wetland researchers. Allowances for normal contingencies involved in work of this type are not included. Getter et al. (1984) suggest that these times should be doubled. The high figures reported in Seneca and Broome (1982) represent plantings of local species in France and may not represent conditions in the United States. Broome et al. (1988) suggest that the reported figures based on the work of Garbisch (150 hours for planting based on 0.6 meter spacing) may be most representative since they are based on the experience of a commercial firm with extensive experience in wetland restoration projects.

The data from U.S. Army Corps of Engineers (1978) suggest that for most common saltmarsh species, planting and digging requires somewhat over 300 hours for 0.6 meter spacing. However, certain species such as big cordgrass may require somewhat more time. Since planting and digging have about equal time requirements, the U.S. Army Corps of Engineers (1978) data are consistent with the figures reported in Broome et al. (1988).

Exhibit 4.1 Reported labor requirements for saltmarsh planting.

Source	Planting Activity	Person-Hours per Hectare		
		(0.5 meter spacing)	(0.6 meter spacing)	(1.0 meter spacing)
Seneca and Broome, 1982	• Digging and separating Halimione springs	220	152	55
	• Digging and separating Puccinellia plus • Planting halimione or Puccinellia	530 1,000	368 694	133 250
Broome et al., 1988	• Manual digging and separating of S alterniflora	200	139	50
	• Mechanized digging and separating of S. alterrniflora	100	69	25
	• Manual planting of S. alterniflora	200	139	50
	• Mechanized planting of S. alterniflora	100	69	25
	• Seeding			
	♦ Harvesting seed	5 (seed spacing not applicable)		
	♦ Threshing seed	2.5 (seed spacing not applicable)		
	♦ Preparing seedbed and sowing	7.5 (seed spacing not applicable)		
	• Planting springs or potted plants using mechanical auger based on work of Garbisch	221	150	54
	• Fertilizing based on work of Garbish	65	45	16
	• Broadcasting seed based on work of Garbisch	10 (seed spacing not applicable)		
USACOE, 1978 taken from Woodhouse et al., 1972	Collecting and transplanting smooth cordgrass by hand	536	372	134
USACOE, 1978 taken from Dodd and Webb, 1975	• Digging, separating and transplanting			
	♦ Saltmarsh	452	314	113
	♦ Black needlerrush	468	325	117
	♦ Smooth cordgrass	536	372	134
	♦ Big cordgrass	844	586	211
USACOE, 1978	• Rule of thumb for:			
	♦ Transplants and sprigs		100-200	
	♦ Rhizomes, tubers and rootstocks		100-150	
	♦ Seeding		10-40	

To summarize the reported data in the literature, it appears that typical labor requirements for planting of sprigs, plugs, or potted plants (in a large field restoration project) requires about 150 hours per hectare for 0.6 meter spacing. An additional equal amount of time is required for digging, separating, and preparing plants dug from a nearby site if this operation is performed in lieu of purchasing the material from a commercial nursery. Seeding and fertilizing requires about 10 to 40 hours per hectare. If seed is harvested and threshed from a nearby site, an additional 10 to 15 hours may be required.

Exhibit 4.2 summarizes the reported cost figures in the literature for saltmarsh restoration. In this table, the reported figures are adjusted to 1992 dollars using the GNP price deflator. Garbisch (1978) reports the fully-loaded cost for seeding and fertilizing (adjusted to 1992 dollars) as $9,680 per hectare. The full-loaded cost (including travel, overhead, and profit) for mechanical planting (based on 0.6 meter plant spacing) is $29,050 per hectare. Semi-mechanical planting (using a hand auger) was $43,570 per hectare.

Broome et al. (1988) reports more detailed information (based on the work of Garbisch). In this case the fully-loaded cost (in 1992 dollars) is $40,970 per hectare. The figures of Garbisch for planting are based on greenhouse-grown plants. Material costs are a large component of costs and include $18,560 per hectare for potted seedlings. These reportedly cost $0.66 per plant. The material cost of slow release fertilizer applied at the time of planting is $2,410 per hectare. A later application of conventional broadcast fertilizer has a material cost of $482 per hectare.

The direct cost of materials and direct labor is reported at $22,650 per hectare, most of which is for materials. Adding in an allowance for travel to the restoration site, per diem, overhead and profit, and an allowance for a replanting guarantee, raises the cost to $40,970 per hectare. The replanting guarantee assumes that 20 percent of the area will be replanted over the development period.

Jerome (1979) reports lower costs. Adjusting to 1992 dollars, seeding costs are reported to range from $1,320 to $2,240 per hectare. Costs for collecting, transplanting, fertilizing, and maintaining a restored marsh are reported to range from $8,900 to $23,600 per hectare. These figures are lower than those based on the work of Garbisch. Few details are provided on the specifics of the restoration effort referred to by Jerome, but it may not include as extensive an effort as assumed by the Garbisch figures.

Exhibit 4.2 Reported costs for saltmarsh planting.

Source	Cost Item	Dollars Per Hectare (Adjusted to 1992)
Garbisch, 1978	• Loading cost for seeding and fertilizing • Loaded cost for mechanical planting (0.6 meter spacing) • Loaded cost for semi-mechanical planting (0.6 meter spacing)	$9,680 $29,050 $43,570
Broome et al., 1988 (based on information from Garbisch)	• Material costs ♦ Potted seedlings ($0.66/plant at 0.6 meter spacing) ♦ Slow release fertilizer ♦ Conventional broadcast fertilizer Labor and material costs Fully loaded costs including travel, per diem, overhead profit plus 20% for replanting guarantee	 $18,560 $2,410 $482 $22,650 $40,970
Jerome, 1979	Seeding Total costs of restored marsh Collectingjj Transplanting Fertilizing Maintaining	$1,340 - $2,240 $8,900 - $23,600
USACOE, 1978	Commercially grown transplants (0. meter spacing)	$7,700 - $41,000

King (1990) notes the problem that many costs associated with restoration in mitigation projects are much lower than could reasonably be expected if the project were to be "true" (i.e., fully successful) restoration. This is because there is frequently considerable pressure to perform the job at the lowest possible cost, so the results are frequently poor.

U.S. Army Corps of Engineers (1978) reports a range of costs for commercially-grown transplant stock. Based on 0.6 meter spacing the cost of transplants ranges from $7,700 to $41,000.

Synthesizing from the data reported in the literature, it appears that figures in the range of $10,000 per hectare are reasonable for a quality seeding effort. Costs in the range of $30,000 to $45,000 per hectare are reasonable using greenhouse-grown transplant stock. Costs on any individual project are highly variable. Some of these key cost variables include:

- The method of marsh establishment (seeding or transplanting);
- The plant spacing in the case of transplanting; and
- Whether or not greenhouse-grown nursery stock is used.

4.2.1.1.3.3 Supplementary Erosion Control Structures

Woodhouse (1979) presents some costs for temporary protection from erosion. Slat-type sand fence costs $3.90 to $5.40 (in 1992 dollars) per linear meter of protection for materials including posts and braces. Installation labor was estimated to be 0.1 person hours per meter in addition to the material costs. Woodhouse (1979) also states that the least expensive sandbag devices cost less than $20 per linear meter.

U.S. Army Corps of Engineers (1978) presents the costs of erosion control structures originally developed for wetland creation on dredged material in exposed locations. The cost of heavy sandbag dikes per linear meter (1992 dollars) were presented as:

- 1.5 meters above bottom - $453; and
- 3.0 meters above bottom - $1,617

4.2.1.1.3.4 Sediment Removal/Replacement

Krebs and Tanner (1981) report on the costs of sediment removal in conjunction with an oil discharge in the Potomac River near the mouth of the Chesapeake Bay. Removal was accomplished by a track-mounted Gradall tractor with a one cubic yard bucket. The stripped area was in a narrow fringing saltmarsh. The substrate was stripped to a depth of 20 centimeters.

Costs per square meter of stripped area (in 1992 dollars) are as follows:

Removal	$3.75
Disposal of stripped material	$2.27
Subtotal for removal alone	$6.02
Backfilling	$1.91
Total including backfilling	$7.93

Plant propagation costs would be additive to the above figures. Replanting costs would be similar to those in the discussion under Replanting.

4.2.1.1.3.5 Vegetation cropping

American Petroleum Institute (1991) provides estimates of the cost requirements for vegetation cropping. These are professional estimates based upon a composite of actual instances of vegetative cropping after oil discharges. The reported cost estimate is based on the following. It was assumed that a four person crew could crop 585 square meters of vegetation per day. The crew was provided with two small boats. The cost per day was estimated as follows:

• Labor - 32 person-hours	@ $35 per hour	$1,120
• Boats - 2 boats	@ $100 per day	200
• Other - 40 percent of the labor and boat costs for miscellaneous equipment and supplies, disposal of oiled debris, and contingency		528
• Total Cost Per Day		$1,848

Under the assumption that this crew could crop 585 square meters per day, the estimated cost for vegetative cropping was $3.16 per square meter.

4.2.1.1.3.6 New Saltmarsh Creation

Costs of creating a new replacement saltmarsh are highly variable. Efforts can range from simple breaching of a dike to inundating a previously drained area to extensively planned efforts involving considerable site excavation. The costs are extremely site sensitive. Every degraded saltmarsh has unique features that pose a challenge to the design of a created saltmarsh.

Factors that can affect the costs include those related to the specific characteristics of the site as well as the features that the designers may wish to include. A partial list of these may include:

- The basic wetland creation method that is being employed (i.e., dike breaching, dredge spoil disposal, etc.);
- The costs involved with acquiring a site. Some sites may have public status and acquired for free while other sites may still be privately owned;
- The costs of the specific design. As with consumer products, the "quality" can range from basic to deluxe. In one project, planning for a specific site resulted in engineering cost estimates that ranged from $5 to 28 million depending on the action that was being considered;
- The amount of excavation work that is required to bring the substrate to the proper level with respect to the tide. When soils are drained, they subside over time;
- The degree of channelization within the designed marsh area. This relates to the degree of intricacy incorporated into the design;
- The distance to a source of water for the tidal flows;
- The topography that must be crossed by the access channels or conduits on their way to the source of tidal water;
- The tidal characteristics at the site;
- The number of dike breaches, conduits, or tide gates that will be required to obtain an appropriate tidal flow;

- The degree to which it is necessary to remove contamination from the site. Many degraded saltmarshes are in areas that are seriously affected by urban, agricultural, or industrial development;

- The amount of litter and debris that must be removed from the site;

- The degree to which it is necessary to remove remnants of old buildings, equipment or other forms of development. Future restoration of a 100 hectare urban saltmarsh in Los Angeles is estimated to cost $10 to $50 million because a major roadway must be relocated (National Research Council, 1992);

- The degree to which materials must be added to the soil to yield the desired characteristics;

- The method of vegetation (i.e., natural, seeding, transplanting, etc.);

- The spacing of vegetation within planted areas;

- The proportion of planted, unplanted, and open water areas within the overall design; and

- The efforts that are required to control exotic species.

Exhibit 4.3 summarizes some costs of typical wetland creation projects. The reported costs for these projects have been adjusted to 1992 dollars. The cost figures presented here are for projects that had either no or minimal planting. Thus, planting costs would be in addition to the costs presented here.

The costs presented in Exhibit 4.3 ranged from $485 to over $70,000 per hectare. The costs presented here have a wide range yet they not represent the extremes. Restoring some wetlands in urban areas was estimated to cost considerably more. The actual cost estimates for a specific site would have to be based on preliminary engineering and biological site surveys.

Exhibit 4.3 Typical wetland creation costs.

Reference Number	Size (hectares)	Cost (1,000 dollars per hectare)	Restoration Method
1	235	843	Dredge disposal with minimal planting
2	272	6,490	Dredge disposal and excavation with minimal planting
3	309	6,480	Dredge disposal and excavation with minimal planting
4	40	9,424	Dredge disposal with minimal planting
5	371	2,429	Tidal gates and regrading
6	91	5,211	Excavation and dike breaching
7	247	17,414	Dredge disposal
8	556	2,675	Excavation and dike breaching
9	32	21,607	Excavation and tidal gate
10	15	18,237	Excavation and dike breaching
11	37	12,010	Excavation; flooding problems encountered
12	383	1,722	Excavation and water control structures
13	543	485	Minimal excavation
14	8	70,560	Dredge disposal and protective structures
15	8	41,160	Dredge disposal
16	5	17,854	Dredge disposal
17	2,470	2,247	Excavation and tidal gate
18	2,470	12,583	Excavation and tidal gate
19	287	27,119	Excavation and tidal gate

Source: Josselyn, 1982; Army Corps of Engineers, 1978; Josselyn et. al. 1991; Purcell and Johnson, 1991.

4.2.1.1.3.7 Low Pressure Flushing

American Petroleum Institute (1991) provides cost estimates for flushing in a marsh including recovery of oil. The cost per hectare was estimated as follows:

Labor - 247 person-hours @ $35 per hour	$ 8,645
Equipment and supplies (10% of labor)	865
Contingency - additional 20 percent	1,902
Total cost per hectare	11,412

4.2.1.1.3.8 Bioremediation

See Section 4.2.6.1.3.5 for a discussion of bioremediation costs.

4.2.1.2 Mangrove Swamp

This section presents a review of the costs associated with each technically feasible restoration option discussed in Section 2.2.1.2 for affected mangrove habitats. These restoration actions include the following:

- Natural Recovery;
- Selected Replanting;
- Construction of Channels for Flushing; and
- Low Pressure Flushing.

4.2.1.2.1 Oil Related Literature

Based on a review of existing literature on techniques for mangrove restoration due to oil related injury, the actions identified above were discussed as technically feasible. With the exception of the channel opening each action was demonstrated in previous restoration projects (Ballou and Lewis, 1989; Getter et al., 1984; Goforth and Thomas, 1979; Lewis, 1979; Lewis, 1981; Lewis, 1990; Teas, 1981; Teas et al., 1989; Thorhaug, 1989).

4.2.1.2.2 Non-oil Related Literature

Injury also occurs to mangrove habitats from natural occurrences and non-oil man-induced impacts. Costs related to the restoration of habitats altered by such impacts are reported in the literature and describe projects which employ different replanting techniques. These historical mangrove restoration projects include those discussed by Teas (1979), Goforth and Thomas (1980), and Sosnow (1986).

4.2.1.2.3 Costs of Restoration Actions

4.2.1.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.1.2.3.2 Replanting

The types of plant material commonly used for mangrove restoration include mangrove propagules, seedlings, or young mangrove trees. The replanting process is typically performed manually using planting material which is either purchased from local nurseries or collected from a healthy adjacent mangrove area.

Unit costs of mangrove restoration projects were derived from several sources which demonstrate alternative planting techniques from mangrove restoration projects that used different types of planting material. As shown in Exhibit 4.4, these sources provide cost estimates for the purchase and/or gathering of planting material and the planting of mangrove propagules, seedlings, and young mangrove trees. The cost sources were derived in different years and thus adjusted using the GNP price inflator to reflect costs in mid-1992 dollars.

4.2.1.2.3.2.1 Propagules

Mangrove propagules are fresh seeds picked from mature fruits on trees in an established mangrove community. Propagules can also be collected from shorelines, but must exhibit characteristics of a propagule recently released from a mature fruit.

The estimated unit costs associated with replenishing mangroves through propagule dispersion for oil-injured habitats using collected or purchased propagules were extracted from various sources in the literature. Lewis (1981) and Thorhaug (1989) both report costs from historical restoration projects where propagules were planted for mangrove restoration. In these oil related restoration projects (Lewis (1979); and Mangrove Systems, Inc. (1980)), collected propagules for planting were used for restoration of injured red mangrove trees. When propagules are collected, they are generally picked from young buds on mangrove trees or collected from the shoreline. Costs for performing the mangrove transplants using collected

Exhibit 4.4 Reported costs for mangrove restoration (\$/ha (\$/m^2) in mid-1992 dollars).

Mangrove Type and Planting Technique	Spacing (m)			Source
	0.30	**0.61**	**0.91-1.23**	
Seeds (collected) Red Mangroves	\$21,376 (\$2.14)	\$5,190 (\$0.52)	\$2,395 (\$0.24)	Teas, 1977 (In: Lewis, 1981; Thorhaug, 1989)
	- - -	- - -	\$11,301 (\$1.13) \$22,604[1] (\$2.26)	Lewis, 1979 (In: Lewis, 1981; Thorhaug, 1989)
	\$24,712 (\$2.47)	\$5,272 (\$0.53)	\$2,471 (\$0.25)	Mangrove Sys. Inc., 1980 (In: Lewis, 1981)
Seeds (purchased) Red Mangroves	\$23,637 (\$2.36)	\$5,761 (\$0.58)	\$2,649 (\$0.26)	Teas, 1977 (In: Lewis, 1981; Thorhaug, 1989)
	\$26,359 (\$2.64)	\$5,766 (\$0.58)	\$2,636 (\$0.26)	Mangrove Sys. Inc., 1980 (In: Lewis, 1981)
Seedlings (purchased) Red, Black and White Mangroves	\$47,059 (\$4.71)	\$11,345 (\$1.13)	\$5,273 (\$0.53)	Teas, 1977 (In: Lewis, 1981; Thorhaug, 1989)
	\$177,254 (\$17.73)	\$44,863 (\$4.49)	\$19,939 (\$1.99)	Mangrove Sys. Inc., 1980 (In: Lewis, 1981)
Seedlings Red Mangroves	- - -	- - -	\$42,328 (\$4.23)	Sosnow, 1981
3-Year-Old Trees (purchased) Red, Black and White Mangroves	- - -	- - -	\$454,055 (\$45.40)	Teas, 1977 (In: Lewis, 1981; Thorhaug, 1989)
	- - -	- - -	\$115,321 (\$11.53)	Mangrove Sys. Inc., 1980 (In: Lewis, 1981)
3-Year-Old Trees (transplanted) Red Mangroves	- - -	- - -	\$82,072 (\$8.21)	Goforth and Thomas, 1979 (In: Lewis, 1981)

[1] Actual cost of a full-scale commercial restoration project.

propagules were reported in Mangrove Systems, Inc. (1980) and differ based on the spacing requirements for various planting scenarios. Restoration costs for propagule planting using collected seeds range from $2.47 per square meter for 0.3 meter spacing to $0.25 per square meter for approximately 1.0 meter spacing (in mid-1992 dollars). In Lewis (1981), collected red mangrove propagules were transplanted at a cost of $1.13 per square meter for 1.0 meter spacing. Lewis (1981) also reports costs for mangrove restoration using purchased propagules for transplant projects.

Restoration costs using purchased propagules range from $2.64 per square meter for 0.3 meter spacing to $0.26 per square meter for 1.0 meter spacing. The estimated unit costs associated with replenishing mangroves through propagule dispersion for non-oil injured habitats, using collected or purchased propagules, were extracted from various sources in the literature. Teas (1977) presents the costs of propagule planting for different spacing requirements, and estimates costs of collected propagules to range from $0.24 per square meter for plants spaced at about 1.0 meter to approximately $2.14 per square meter for 0.3 meter spacing. Lewis (1981) and Thorhaug (1989) both report similar costs. Restoration costs for propagule planting using purchased seeds range from $0.26 per square meter for 1.0 meter spacing to $2.36 per square meter for 0.3 meter spacing. Restoration costs using mangrove propagules are summarized for each mangrove type and planting action in Exhibit 4.4.

4.2.1.2.3.2.2 Seedlings

Mangrove seedlings used as transplant material are generally grown to a specific size or age in nursery conditions before being planted. The growth of fresh seeds to an average height of 0.5 meters with some leaves present is generally considered suitable for planting material. Seedlings with these physical characteristics range from 6 to 18 months in age.

The literature on oil related mangrove restoration, found in Lewis (1981), reports costs from one restoration project where 6-month old red, black, and white mangrove seedlings were used for mangrove transplants (Mangrove Systems, Inc., 1980). The cost of restoration projects using purchased seedlings range from $17.73 per square meter for 0.3 meter spaced transplants to $1.99 per square meter for 1.0 meter spacing.

Costs for non-oil related mangrove seedling transplants are summarized below. Teas (1977) also reports costs for seedling transplants, estimated to range from $0.53 per square meter for 1.0 meter spacing to $4.71 per square meter for planting seedlings at 0.3 meter spacing. Other reported costs for seedling plantings for mangrove restoration due to dredging impacts include those presented by Sosnow (1986). Costs were derived for a full scale pilot restoration project in which 0.32 hectares of mangroves were restored. Adjusted to reflect costs for one hectare of mangrove habitat, project costs were estimated to total $42,328 or $4.23 per square meter of area restored. These costs reflect all costs associated with obtaining plant material and planting activities, land preparation, and rip-rap replacement (Sosnow, 1986). Costs for seedling transplants are also summarized in Exhibit 4.4.

4.2.1.2.3.2.3 Young Mangrove Trees

Young mangrove trees are generally grown in nurseries to approximately 1.0 meter in height and provide more rapid growth as transplant material to help with substrate stabilization. Use of 1.0 meter trees is more costly, as documented by past restoration experiments, but survival of transplants is generally greater, especially under stressful habitat conditions such as increased wave energy. Two studies in the literature tracked the costs of past mangrove restoration projects which used young trees for transplant material. Lewis (1981) summarizes the work of Mangrove Systems, Inc. (1980) and identifies the costs of the respective mangrove restoration projects. Restoration costs for mangrove restoration using three-year-old trees as transplants (at 1.0 meter spacing) were reported to be approximately $11.53 per square meter. A more recent analysis of mangrove restoration criteria notes that direct restoration of injured large mangrove trees with nursery-raised replacements is "prohibitively expensive." Costs to grow a single red mangrove tree to 5 meters in height (4 square meter coverage) including installation were estimated to be in excess of $11,000 (Crewz and Lewis, 1991). The higher cost of transplanting older, more mature mangrove trees relative to propagules and seedlings may be a result of the decreased availability of suitable donor trees and high costs of nursery supplied plant material.

Mangrove restoration using young mangrove trees (one to three years old) for transplant material was also been documented in the literature as suitable material for long-term restoration success for non-oil related injury (Goforth and Thomas, 1979; Teas, 1977). Two studies in the literature tracked the costs of past mangrove restoration projects (non-oil related) that used young trees for transplant material. Teas (1977) estimates costs of nursery-grown three-year-old trees to be nearly $74.00 each (adjusted for inflation). The costs for planting these trees at a spacing of 1.23 meters is estimated to cost approximately $454,000 per hectare, or $45.40 per square meter. Goforth and Thomas (1979), as reported in Lewis (1981), estimated costs of planting small mangrove trees for shoreline stabilization to be $82,000 per hectare, or approximately $8.20 per square meter. The costs for mangrove restoration using young tree transplants are also summarized in Exhibit 4.4.

It is important to note that all costs reported for each type of restoration project were incurred during small-scale restoration experiments where associated costs for profit and overhead were excluded. Depending on what plant material is used, a balance must be struck between cost, expected success, and time lapse until the planting is mature. The cost of replanting varies depending on the plant material used and spacing of the installations. It is apparent that for a given spacing distance, the costs increase substantially from the lower end (using propagules) to the higher end (using larger trees). For this reason, spacing is a critical factor when planning a restoration project. Reducing the spacing by a distance of one-third (from 0.91 meter to 0.61 meter), for example, more than doubles the number of installations required.

In addition, the need for other cost-generating components in a restoration project, such as surveys, meetings with regulatory agencies, and travel, may increase the costs of the restoration project considerably. Lewis (1979) identifies costs associated with a full-scale commercial restoration project where these components were included in the total cost of the project. In this case, the estimated cost per square meter of mangrove habitat restored (using collected propagules) was reported to be $2.26 per square meter, twice the costs of propagule planting alone.

For any completed replanting operation, a monitoring program should be developed in order to track the progress and reliability of habitat restoration. There were no reported costs in the literature for monitoring programs associated with any of the documented mangrove restoration projects. However, cost estimates of a generic monitoring program can be found in Section 4.4.

4.2.1.2.3.3 Construction of Channels for Flushing

No cost data were reported in the literature for mangrove restoration involving the opening and flushing of channels to circulate and dilute remaining concentrations of the pollutant. However, engineered estimates can be derived based on the expected costs of activities that comprise this restoration action. Cost estimates could be derived based on the level of effort necessary to excavate a designated mangrove habitat. These estimates would be based on such factors as required labor, materials, and equipment mobilization/demobilization.

According to Ballou and Lewis (1989), excavation of channels into an affected area would be a relatively expensive and complex task compared to other restoration actions (i.e., natural recovery and replanting).

4.2.1.2.3.4 Low Pressure Flushing

See Section 4.2.1.1.3.7.

4.2.2 Freshwater Wetlands

This section is divided into the emergent shrub and forested wetlands.

4.2.2.1 Emergent Freshwater Wetlands

Section 2.2.2.1 discusses the technical feasibility of emergent wetland restoration actions, which include the following:

- Natural Recovery;
- Replanting;
- Sediment Removal/Replacement;
- Vegetation cropping;
- New Wetland Creation; and
- Low Pressure Flushing.

Costs of the actions are discussed in the following sections.

4.2.2.1.1 Oil Related Literature

Although two reports addressing wetlands restoration following oil discharges were identified in Section 2.2.2.1.1 (Foley and Tresidder, 1977; and Pimentell, 1985), no information was identified regarding costs or economics of emergent wetlands restoration efforts in response to an oil discharge. Much of the literature regarding saltmarsh restoration may be directly applicable (see Section 3.2.1.1), although it should be noted that freshwater marsh plant diversity is typically much higher than that of saltmarshes, adding to the complexity, and therefore cost, of accomplishing a successful restoration.

4.2.2.1.2 Non-oil Related Literature

No information was identified regarding costs of restoration of emergent wetlands following discharges of hazardous materials. The following information on restoration costs is related to creation of wetlands, primarily on old mine lands, dredge disposal areas, or previously drained wetland being returned to marshland from agriculture.

4.2.2.1.3 Costs of Restoration Actions

4.2.2.1.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.2.1.3.2 Replanting

Landin (1990) in a discussion of emergent wetland creation states that "marsh propagation costs will be determined by the labor and expense of obtaining, transporting, handling, storing, and planting propagules, the number of propagules required, any soil treatments necessary such a fertilization, and maintenance efforts."

Labor requirements for replanting - Labor costs for general (no specific species) planting activities are shown below. Additional labor costs are presented in Section 4.2.1.1.3.2, regarding replanting saltmarshes.

Source	**Activity (at 0.9 meter centers/intervals)**	**Person hours per hectare**
Landin, 1990	Digging, preparing, and planting transplants	98.7-198
Landin, 1990	Landin, 1990 Transplanting rhizomes, tubers, and rootstock	98.7-148
Landin, 1990	Seeding	10-40
Ternyik, 1978	Digging and planting "Tufted Hairgrass" in sandy material using professional nursery work force	86
Knutson, 1977	Excavating, separating, and planting "sprigs"	123
Knutson, 1977	Prepare and plant plugs	1,111
Knutson, 1977	Seeding, including "harvest, storage, dispersal	62
Dodd, Webb, 1975	Hand dig, separate, and transplant propagules	111-289

Landin (1982) reported that activities requiring plantings at 0.9 meter (1 yard) intervals will require roughly 9,900 plants per hectare. A 0.45 meter spacing will require four times as many plants per hectare, and, therefore, four times the labor. Plants set at 1.8 meters will require half the number of plants per hectare, and, therefore, half the labor hours. Values in the table above may be adjusted appropriately. Landin also points out the following:

Marsh propagation costs will be extremely site specific and will reflect such factors as logistics, man-hours costs, efficiency, plant design, and the texture of the substrate. The data reported here (see Landin, 1982, estimates above) are developed from sites that could support conventional equipment. Should the substrate of the site be poorly consolidated fine-textured material, more person power will be required to propagate due to "trafficability problems."

Estimates of total planting costs - Lee et al. (1976) estimated the cost of establishing two month old plant species at a density of 12,400 per hectare to be the following:

Type of vegetation	**Range of costs (1992 dollars)**	**Average cost**
Naturally available vegetation	$10,927 to $13,826	12,376
Commercially available vegetation	$15,387 to $18,286	$16,836

Based on these estimates, costs for using commercially available vegetation will run approximately 36 percent higher than vegetation available naturally for transplant.

Costs of Individual Plants - Marsh plants may be purchased at nurseries, although availability of many species may be limited. Some costs noted are as follows.

Source	Description	Per plant cost (1992)
Crabtree et al. (1990)	Purchasing and planting marsh plants	$0.98
Lee et al. (1976)	Purchasing and planting commercially available marsh plants	$3.03 (1)
Korschgen (1988)	Purchase price, winter buds of Wild celery	$0.12
Landin (1982)	Marsh plant purchase	$0.20 to 1.06

(1) Note that the value for commercial plants in 1976 was $1.36 per plant indexed to 1992 dollars to be $3.03. This calculation is made assuming that the value has moved only in response to inflation. In fact, the supply of commercially available marsh plants may have increased dramatically since 1976 in response to increased wetlands creation/restoration activities, in the process depressing prices in spite of inflation.

Using Muck as a Substrate and Seed Bank - Brown, Gross, and Higman (1984) studied the feasibility of placing peat as a substrate prior to revegetating as part of creating a wetland. As part of the study, they maintained detailed records of the number of loads, estimated volumes per load, round-trip travel time, and hours of equipment operation. Costs for two different peat placement operations are presented below.

Site: Activity	Machinery Used	Cost per m^3 (1992 dollars)
Site 1: Digging and transport	Cat 627 pans, D-8 Dozer, Motor Grader	$13.76
Site 1: Spreading	Komatsu Dozer	$1.89
Site 2: Digging and transport	Dragline, Payloader, and Dump truck	$2.28
Site 2: Spreading	D-3,D-5,D-6 Dozers	$4.52

The first site used Cat 627 scrapper pans to dig and transport the peat material and a dozer to spread the peat. At the second site, a dragline was used to excavate the material, 7.65 cubic meter (10 cubic yard) dump trucks were used for transport, and several large dozers were used to spread the peat. The general presumption by the authors was that small equipment may be more efficient than large equipment for this type of operation.

The cost for the acquisition of the peat is not included in the study. The peat reportedly was removed from "donor" swamps in forested wetlands that were to be mined. Obviously in cases where a donor site is not readily available for access to free peat, the cost of purchasing peat for transport to the site must be included.

The cost of incorporating a muck layer into the substrate (as described by Bacchus, 1989) were not broken out to allow an estimate of the cost of digging and transporting muck versus cost for grading, planting trees and herbaceous species, and project management. Cost per hectare were approximately $96 thousand per hectare for the entire restoration (1992 dollars).

4.2.2.1.3.3 Sediment Removal/Replacement

Sediment Removal - No cost estimates were identified in the literature regarding the cost of sediment removal from freshwater emergent wetlands. Krebs and Tanner (1981) reported the costs of sediment removal following impacts of an oil discharge on a saltmarsh near the mouth of the Potomac River. The costs of removal were estimated at $6.02 (1992 dollars) per square meter for removal and backfilling costs of $1.91 for a total of $7.93 (see Section 4.2.1.1.3.4 for details).

Crabtree et al. (1990) reported the costs of "spreading topsoil," or mulching as the activity is commonly described. The activity is typically employed for replanting purposes as the mulch is full of seeds, roots, and rhizomes. The unit costs are presented below.

Activity	**Cost (1992 dollars)**
Cost of excavating topsoil	$5.70 per cubic meter
Cost of spreading topsoil	$3.05 per square meter
Total cost for 15.2 centimeter (6 inch) layer	$3.84 per square meter

4.2.2.1.3.4 Vegetation Cropping

Foley and Tresidder (1977) and Pimentell (1985) both reported on the technical feasibility of cropping vegetation. Neither report, however, discussed the costs of this type of operation. The American Petroleum Institute (1991) estimates the costs of vegetative cropping of saltmarshes at $3.16 per square meter (costs and assumptions are described in Section 4.2.1.1.3.5).

4.2.2.1.3.5. New Wetland Creation

Diking/plugging drains - Reclaiming previously drained wetlands in many cases is a simple matter uf plugging the fixture that was installed to drain the water off the area (Piehl, 1986; Rondeau, 1986; Kusler, 1986). Kusler et al. (1986) reported that 'the cost to restore small wetlands with a single dike averages' $373 per wetland (1992 dollars). On these wetlands the drain is diked with a plug on average that is "4 feet high, 10 feet wide, and 45 feet long with the face of the dike reshaped to the contours of the wetland basin." On larger wetlands areas, water control structures are incorporated into the dike, the costs for these range from $1,864 to $12,430 each (1992 dollars). Rondeau (1986) reported plugging a tile and ditch drainage structure for a cost of $323 (1992 dollars). At a different area the author discussed the creation of seven wetland basins for an average cost of $430 per wetland (1992 dollars). The author reported maintenance costs (actually costs paid to farmers to conduct weed control) of $235 per hectare per year. Piehl (1986) reported similar annual expenditures of $222 per hectare (1992 dollars).

Examples of Total Costs - Total costs for selected freshwater emergent wetland restoration/creation operations are as follows:

Source and Project	Activities	Cost per hectare (1992 dollars)
Crabtree et al. (1990); (French Creek)	Cleared, excavated, graded, returned topsoil, planted with nursery stock	$23,578
Lee et al. (1976)	Planted naturally available vegetation at 12,400 plants per hectare (no construction costs)	$12,376
Lee et al. (1976)	Planted commercially available vegetation at 12,400 plants per hectare (no construction costs)	$16,836
Crabtree et al. (1990)	Total cost for mulching using a 15.2 centimeter (6 inch) layer (no grading or planting costs are included)	$38,370
Crabtree et al. (1990) (Rancocas Creek)	Revegetated by planting (construction costs not included)	$28,908

4.2.2.1.3.6 Low Pressure Flushing

See Section 4.2.1.1.3.7.

4.2.2.2 Scrub-Shrub Wetland

No cost data for restoration of scrub-shrub wetlands was identified. Costs of equivalent emergent and/or forested wetlands (i.e., same species composition) give an indication of the potential costs.

4.2.2.3 Forested Wetlands

Section 2.2.2.3 discusses the technical feasibility of forested wetland restoration actions including the following:

- Natural recovery;
- Replanting; and
- Forested Wetland Creation.

4.2.2.3.1 Oil Related Literature

No information was identified on restoration efforts in response to an oil discharge.

4.2.2.3.2 Non-oil Related Literature

Little was published addressing restoration of forested wetland. The following information on restoration costs is related to creation of wetlands, primarily on old mine lands or dredge disposal areas.

4.2.2.3.3 Costs of Restoration Actions

4.2.2.3.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.2.3.3.2 Replanting

In practice, replanting in response to discharge impacts may follow vegetation removal or soil removal/replacement. In the literature no discussion of remediating discharge impacts on forested wetlands was noted. The cost discussion below is related to replanting of trees during the creation of wetlands. These costs are referenced in New Wetland Creation, but should be applicable to replanting as part of a mitigation restoration effort.

Transplanting Seedlings - Hammer (1992), in his book Creating Freshwater Wetlands, discussed the costs of nursery grown plants as reported below.

Type of Nursery grown plants	Cost per plant (1992 dollars)	Comments
Potted materials	$1.00 to $3.00	Easily planted, suffer less transport and planting shock
Bare-root seedlings	$0.25 (or less)	Susceptible to transport planting shock
Containerized seedlings (in trays, molded peat or wood cups)	$0.50-$0.75	May survive in sites too harsh for bare-root seedlings
Bagged (root-ball) saplings	$3.00 to $50.00	Less susceptible to transport planting shock

These costs do not reflect costs of planting or maintaining the seedlings/saplings or preparing the substrate.

Landin (1982), in discussing the creation of a wetland on a dredge disposal site in Texas, noted that generally, 111 to 222 person hours per hectare should be allowed for "digging, preparing, and planting transplants on a site." The author noted that the habitat development (planting) aspects of the Trinity River project could be carried out for about $5,323 per hectare (1992 dollars). This reportedly entailed planting trees at 3.05 meter (10 foot) intervals and herbaceous species at 0.9 (3 foot) meter centers. The costs do not include, however, diking of the dredge spoil area or any contouring or other ground moving.

Weston and Brice (1991) reported on replanting of indigenous species following removal of exotic pest species. The 1.0 hectare swamp area and 0.2 hectare ponded area were planted with 228 trees and shrubs. The trees, purchased at a local, native-plant nursery, were planted in 3-5 gallon root balls while shrubs were planted in one gallon root balls. Total costs for the plants (both trees and shrubs) were $1,243; the labor used was from a non-profit organization at an hourly rate of $9.82 per hour (1992 dollars).

Denton (1990) in a report regarding using cyprus at forested mitigation sites, estimated costs for installation, monitoring, and maintenance of forest mitigation areas. The costs were reported as $5.32 for small trees with roots filling a 1-gallon can, $7.45 for 3-gallon trees, and $23.41 for 7-gallon trees (1992 dollars). Maintenance costs were estimated at $6,567 per hectare for the first two years, $4,728 per hectare the third year, and $3,153 per hectare thereafter (1992 dollars). The author estimated monitoring costs at $2,890 per hectare per year. In estimating total costs the author took into account differences in planting small versus large plants, their cost differences, and, assuming a goal of 33 percent canopy was to be achieved, calculated costs and time to achieve canopy cover. The author estimated that "planting small trees (1-gallon) densely at 2,470 per hectare and monitoring until a 33 percent canopy cover is attained will require 5 years" at an estimated cost of $51,753 per hectare in 1992 dollars. The author contrasted that strategy by estimating that planting larger (7-gallon) trees less densely at 990 per hectare (400 per acre) will require 7.5 years at an estimated cost of $76,844 per hectare in 1992 dollars. The author concludes that the former strategy is cheaper, faster, and a good opportunity to "create a wetland with trees as dense as those found in many natural wetland systems."

<u>Transplanting With a Tree Spade</u> - Posey et al. (1984) reported the technical feasibility of replanting adult trees 9 meters or less in height. The trees were transplanted using a Big John 78 Tree Spade capable of handling a 3400 kilogram ball 2 meters in diameter. Accurate records were reportedly kept indicating sizes and quantities transplanted as well as survival rates and regeneration statistics. The cost of this spade transplant was $72 to $85 per tree or approximately $9,847 per hectare in 1992 dollars (no planting density was provided). No cost was included for purchase of the trees themselves as the trees were on a plot of land scheduled to be cleared for strip mining. Obviously in cases where a donor site is not readily available for access to free trees, the cost of purchasing trees for transport to the site must be included.

<u>Transplanting Using Boxing</u> - Carothers et al. (1990), in an appendix to their study on restoration of riparian lands, noted that "mature trees of any size can be boxed and moved." They note that while this action was used to salvage trees in areas to be developed, the action has not been used in restoration or creation projects. They state that "in some cases this technique (action) may be useful," but they note that its cost is "its main drawback." The authors' estimates were reportedly $532 to $1,064 per tree in 1992 dollars. They discussed one case in Arizona where 240 trees were salvaged (boxed and removed) from a 13 acre (5.27 hectare) riparian woodland. They reported the costs as high, ranging from $53.20 to $85.12 per basal diameter inch (2.54 centimeters) (1992 dollars) with an "additional 40 percent of this cost required to replant and maintain in a nursery."

<u>Revegetation Using Cuttings</u> - One case study on a wetland created in Idaho reported that ten to twenty of the unrooted cuttings reportedly could be planted with the same effort and expense of one rooted cutting (Jensen and Platts, 1990). However, survival rates of unrooted cuttings would be much lower, so that the total expenses of replanting would not be as low as this suggests.

<u>Revegetation Using Seeds</u> - In a report discussing seeding with oak acorns, Johnson and Krinard (1987) collected information regarding cost, seed handling, planting methods, survival, growth, and competition of using acorns to revegetate with oaks. They found the costs for the collection of acorns to be $59.51 per hectare ($24.10 per acre) planted. The authors, estimated an additional cost of $5.95 to $14.89 per hectare ($2.41 to $6.03 per acre) if the seeds were stored one year (1992 dollars). They further noted that "total cost of establishment by direct seeding, including collection and handling of seeds, labor, and site preparation," may range from $35.70 to $148.77 per hectare ($14.46 to $60.25 per acre) in 1992 dollars.

McElwee (1965) discussed the advantages and disadvantages of direct seeding of hardwoods in river bottoms. In terms of cost, the author indicated cost savings of 25 to 33 percent over hand planting seedlings, which, the author notes, is often required because the saturated soils prohibit the use of mechanical planting.

4.2.2.3.3.3 Forested Wetland Creation

Most of the costs related in the previous actions were incurred or estimated during the creation of new wetlands. Included below are several restoration projects for which costs were not broken-out by the various aspects of the effort (i.e., replanting versus ground preparation).

One wetland creation project in central Florida was characterized by the inclusion of a muck layer into the substrate followed by planting with 10 species of trees and 9 herbaceous species (Bacchus and Webb, 1989)). The cost of the entire wetland creation project was $493,742 for the 5.1 hectare area, or $96,812 per hectare in 1992 dollars.

Haynes and Crabill (1984) reported that from 90 to 95 percent of the total cost of reclaiming 6.5 hectares of phosphate-mined lands as forested wetlands were for "earthmoving work involving heavy equipment." Revegetation of the project site was estimated to be 2-3 percent of the total reclamation cost while monitoring was 1-2 percent of the total cost.

In practice, dead trees and other affected vegetation may be removed to reduce the oil or other hazardous material residue remaining on the vegetation. In the literature, no discussion of remediating discharge impacts on forested wetlands was noted. Weston and Brice (1991) reported on the technical feasibility of removing exotic, unwanted species from a wetland site prior to replanting with indigenous species. The one hectare "low swampy area" required 664 hours of labor for a total of $6,576 (1992 dollars) to cut the trees with a chain saw and haul the trees out by hand. The labor was supplied by a non-profit organization, a residential treatment facility for adjudicated youth. The authors stated their belief that this use of non-profit labor had "particular applicability to other habitat restoration projects." Tipping fees for hauling the unwanted slash were $3,580 and the cost for chemicals to treat the stumps and unwanted vegetation was $1,585 (1992 dollars). The area restored was one hectare, so by definition, all costs reported are per hectare unit costs. The trees removed in the Weston and Brice study were a small, understory-type species known as Brazilian Pepper. Removal of large trees such as cyprus and red oak would require a full-scale timber operations using mechanical skidders to haul out timber and logging trucks with lift arms to pick up and remove the logs. Costs for these operations were not available in the literature regarding forested wetlands restoration.

4.2.2.4 Bogs and Fens

No cost data for restoration of bogs and fens was identified.

4.2.3 Vegetated Beds

4.2.3.1 Macroalgal Beds (Estuarine and Marine)

4.2.3.1.1 Intertidal Macroalgal Bed

No cost data were identified for replanting of intertidal macroalgal beds. Other restoration actions for this habitat would be as described for rocky or cobble-gravel shores (Sections 4.2.6.1 and 4.2.6.2).

4.2.3.1.2 Kelp Bed

Section 2.2.3.1.2 discusses the technically feasible restoration actions identified for injured kelp bed habitats. These actions include:

- Natural Recovery;
- Replacement with Transplants; and
- Vegetation Cropping.

This evaluation was based on a review of available literature as discussed below.

4.2.3.1.2.1 Oil Related Literature

As discussed in Section 2.2.3.1.2, no literature exists that documents actual restoration activities performed for oil-injured kelp habitats. Similarly, there exists no documented costing information for oil related restoration actions performed for kelp habitats.

4.2.3.1.2.2 Non-oil Related Literature

Actual costing information reported in the literature for restoration actions performed on kelp habitats after non-oil related injury was identified in two cases. The first was reported in a report prepared for the California Department of Fish and Game (CDFG) by Kelco Co., a primary commercial harvester of kelp (CDFG, 1990). Costs reported in this document are related to restoration activities performed on California kelp beds using three restoration techniques: use of artificial growth centers (AGCs), use of AGCs with kelp transplants, and stapling loose plant material to the habitat bottom.

The second case where costing information was reported was found in Shiel and Foster (1991). This paper discusses a range of historical restoration activities performed on kelp habitats and documents only one instance where related costs were reported. The reported costs are those related to the above-mentioned restoration actions performed by Kelco Co., as reported in CDFG (1990).

No literature on the costs of restoring other than *Macrocystis* kelp beds (e.g. *Laminaria*) were found. This section will, thus, only review costs of *Macrocystis* restoration efforts.

4.2.3.1.2.3 Cost of Restoration Actions

4.2.3.1.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.3.1.2.3.2 Replacement with Transplants

Three technically feasible restoration alternatives for restoration of kelp beds include the placement of artificial growth centers (AGCs) into the habitat to induce natural colonization, placement of artificial growth centers with transplants for accelerated recovery, and stapling of loose kelp plants to the habitat bottom to increase the habitat cover. The reported costs for each of these activities separate restoration costs into two primary components. Costs related to the fabrication of materials used and costs of material deployment to the injured habitat. The costs associated with each of these restoration actions are summarized in Exhibit 4.5 and described below.

4.2.3.1.2.3.2.1 Artificial Growth Centers

The use of artificial growth centers as a restoration action involves the placement of "mushroom" anchors on the habitat bottom to act as a surrogate substrate for natural macroalgal spore colonization. The anchors are constructed of concrete with two rebar "handles" set into the flat surface of the anchor to support plant attachment. Mushroom anchors are deployed into the water by the use of small boats or larger vessels, depending on the acreage cover desired for one day's deployment.

Cost estimates developed by Kelco Co. (CDFG, 1990) for a restoration project performed in the Santa Barbara, California area include unit costs for all components of the restoration project, such as the level of effort required for anchor fabrication, anchor deployment, and shipping. These cost components are identified in Exhibit 4.5. Total costs to restore one hectare of kelp bed were estimated at $1,546 adjusted to reflect costs in mid-1992 dollars.

4.2.3.1.2.3.2.2 Artificial Growth Centers with Transplant Material

This action also involves the use of the "mushroom" anchors described above, with attached plant material. The cost components associated with this method are similar to those used for artificial growth centers, with the exception of additional costs for nursery-grown macroalgal plant material. The use of transplant material significantly increases the cost of restoration per unit of habitat restored, as shown in Exhibit 4.5.

The restoration project performed by Kelco Co., as described above (CDFG, 1990), also performed kelp restoration using artificial growth centers with attached transplant material. The costs for this action are higher per acre of coverage due to the increased cost of transplant material and additional labor and materials required to handle and attach the plant material. Total costs for the use of artificial growth centers with transplants were reported to be $3,142 per hectare of kelp habitat restored.

4.2.3.1.2.3.2.3 Staple Loose Plants to Habitat Bottom

The third action for kelp transplants involves the stabilization of loose kelp plants. A demonstrated method of performing this action, as documented by Kelco Co.'s restoration efforts (CDFG, 1990), involves the stapling of loose plants to the habitat bottom through the use of large (two foot long) rebar staples with hose "barbs" attached to the ends. Two staples are used to secure one loose plant. Unlike the vessel deployment of the anchor transplants, the stapling method requires the use of divers. The staple method is generally used in areas where the bottom substrate is soft, such as silt, mud, or sand. The cost components required for this method are summarized in Exhibit 4.5.

Efforts at using staples to stabilize a kelp habitat were based on a restoration project performed in a predominantly sandy environment. Costs for this method were lower than those described for anchors and anchor transplants, estimated at $1,833 per hectare. These costs reflect adjustments to mid-1992 dollars.

As mentioned above, the costs reported above for kelp restoration were identified from two literature sources, a report prepared by Kelco Co. for the CDFG based on actual restoration performed and an academic paper prepared on the status of kelp restoration (Shiel and Foster, 1991). This latter paper reported the costs from the restoration work performed by Kelco Co., thereby describing the same project. Shiel and Foster also noted that the restoration work performed by Kelco was the only documented project where restoration costs were specifically developed. Therefore, the costs reported may not be an accurate representation of a kelp restoration effort, due to the variability often observed in damage assessment. Also, the restoration actions were focused only on the kelp, not on restoring associated plant and animal species.

Exhibit 4.5 Reported costs for kelp bed restoration actions for non-oil-related injury ($/ha in mid-1992 dollars).

Cost Components	Restoration Technique		
	Artificial Growth Centers (AGCs)	**AGCs with Transplants**	**Stapling Loose Plants**
Fabrication:			
Materials	$452	$452	$430
Labor	252	252	42
Facilities	22	22	54
Deployment:			
Vessels	378	795	252
Labor	180	422	909
Travel	30	30	20
Miscellaneous	126	126	126
Harbor	32	64	--
Forklift	20	40	--
Shipping	54	40	--
Transplants	--	855	--
Total Cost	$1,546	$3,142	$1,833

Source: California Department of Fish and Game, 1990.

In addition, the reported costs did not address the costs associated with a monitoring program. For all types of restoration work, a monitoring plan should be developed in order to measure the reliability of the restoration effort. As described above for the natural recovery option, the monitoring program would be designed based on the objectives and standards of environmental recovery, depending on the nature and extent of injury.

4.2.3.1.2.3.3 Vegetation Cropping

No cost data were identified for vegetation cropping of kelp beds.

4.2.3.2 Seagrass Beds

This section provides cost estimates for restoration activities related to seagrass beds.

As identified in Section 2.2.3.2, technically feasible restoration actions for injured seagrass habitats include:

- Natural Recovery; and
- Replanting.

The following sections provide cost estimates of these actions as identified in the literature for historical restoration projects.

4.2.3.2.1 Oil Discharge Related Literature

As discussed in Section 2.2.3.2, there are no documented cases in the literature where seagrass beds were restored due to oil injury.

4.2.3.2.2 Non-oil Related Literature

Cost data for seagrass restoration is documented in several literature sources for restoration projects performed due to injury resulting from activities such as pipeline construction, coastal development, and natural occurrences. The following literature sources identify cost information related to historical seagrass restoration projects where replanting activities were performed: Fonseca et al. (1979); Phillips (1980, 1982); Thorhaug (1980); Fonseca et al. (1982b); Thorhaug (1986); Thorhaug (1989); Thorhaug and Austin (1976); Austin and Thorhaug (1977); and Fonseca et al. (1990b).

4.2.3.2.3 Cost of Restoration Actions

4.2.3.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.3.2.3.2 Replanting

As discussed in Section 2.2.3.2 regarding the technical feasibility of seagrass restoration, several types of planting material can be used to replant injured seagrass habitats. The available cost information from historical replanting projects document cases where plugs and shoots were used as planting material. Exhibit 4.6 summarizes the cost information for each of these replanting techniques and identifies the source of data. The cost of seagrass per hectare ($0.84 per m^2) to $37,800 per hectare ($3.80 per m^2). grass plugs range from approximately $8,352 per hectare ($0.84 per m^2) to $37,800 per hectare ($3.80 per m^2).

Costs for replanting seagrass habitats located in subtropical and tropical habitats are reported in the literature for transplanting activities using seedlings, plugs, and shoots. As discussed in Section 2.2.3.2., the most feasible restoration action for species of seagrass found in the subtropical zone is the use of seeds or seedlings as planting material for eelgrass for which seeds are available or can be collected. Seeding other seagrasses is more problematic since ample quantities of seeds do not appear to be available. Exhibit 4.7 summarizes the reported costs for this method and other techniques used in historical restoration projects. The cost range reported for seedling replanting varies widely, as reported by Thorhaug and Austin (1976). These data are widely reported in other sources as well. Costs for seedling replanting range from approximately $93,337 per hectare ($9.33 per m^2) to nearly $622,277 per hectare ($62.23 per m^2). These estimates reflect the higher end of reported costs, primarily due to the intensive labor and amount of materials required to achieve recovery goals. Thorhaug (1986) reports on another project where restoration using seedlings was performed. These costs were much lower, estimated at $23,150 per hectare ($2.32 per m^2). Only one restoration project reported costs using seagrass plugs, at a cost of $200,232 per hectare ($20.02 per m^2). Another, more recent source reported costs of experimental restoration activities where seagrass shoots were used. These materials were planted using three different methods: the staple, peat pot, and core methods. Costs for these methods were estimated to range from $2.58 per m^2 to $7.52 per m^2 (due to the small size of the project, costs were only reported for the area restored).

Exhibit 4.6 Reported costs for temperate seagrass (Eelgrass) restoration actions ($/ha ($/m^2) in mid-1992 dollars).

Replanting Technique	Cost	Source
Plugs	$37,800 ($3.78)	Robilliard and Porter, 1970 (In: Fonseca et. al. 1979; Thorhaug, 1986; Phillips, 1980, 1982)
	$8,352 ($0.84)	Ranwell et. al. 1973 (In: Fonseca et. al. 1979; Thorhaug, 1986; Thorhaug, 1980; Phillips, 1982)
	$11,713 ($1.17)	Churchill et. al. 1978 (In: Fonseca et. al. 1979; Thorhaug, 1986; Phillips, 1980, 1982)
	$33,464 ($3.35)	Goforth and Peeling, 1979 (In: Thorhaug, 1986)
Shoots with Woven Mesh Anchor Actual Projected	$ 31,771 ($3.18) $10,425 (1.04)	Fonseca et. al. 1979 (Also in: Thorhaug, 1986; Phillips, 1980, 1982)
Shoots	$42,480-$63,720 ($4.25-$6.37)	Fonseca et. al. 1982b

For eelgrass restoration using shoots, estimated costs range from $10,425 per hectare ($1.04 per m^2) to $63,720 per hectare ($6.37 per m^2). These costs reflect adjustments for inflation to show estimated costs in mid-1992 dollars.

Exhibit 4.7 Reported costs for subtropical and tropical seagrass restoration ($/ha ($/m^2) in mid-1992 dollars).

Replanting Technique	Cost	Source
Seedlings	$93,337[2] ($9.33) Cover of 3000 blades/m^2 in 2.5 years	Thorhaug and Austin, 1976; (Also in: Austin and Thorhaug, 1977; Phillips, 1980; Thorhaug, 1980; Thorhaug, 1989)
	$124,452 ($12.45) Cover of 4000 blades/m^2 in 2.5 years	
	$311,139 ($31.11) Cover of 1000 blades/m^2 in 0.8 years	
	$622,277 ($62.23) Cover of 2000 blades/m^2 in 0.8 years	
	$23,150 ($2.32)	Thorhaug, 1986
Plugs	$200,232 ($20.02)	Thorhaug, 1980
Shoots in Test Plots Using:		Fonseca et. al. 1990
Staple	($4.06-$4.40)	
Peat pot	($2.58-$3.18)	
Core	($7.52)	

[2] Costs reflect adjustment for additional cost components.

The range of cost estimates is broad due to several factors. First, planting sites and conditions vary for all types of injury, and restoration requirements can be very different in all cases. Second, the cost of planting seagrass depends on a series of factors including type of labor depth of planting, experience with planting action, type of equipment, accessibility to planting site, and proximity of available donor plant materials. Third, costs can vary based on the size of job and degree of site constraints especially where small-scale and large-scale restoration projects are performed. Costs should take into account the scale of operation. In addition, some costs may or may not include costs for transportation and other costs (i.e., insurance, payroll, administrative overhead, profit, etc.). As a result of these conditions and the variability of site restoration attempts, caution should be taken in using these estimates to determine future restoration costs.

4.2.3.3 Freshwater Aquatic Beds (Submerged and Floating Vegetation)

No cost data were identified for restoration of freshwater aquatic beds.

4.2.4 Mollusc (Oyster) Reef

Section 2.2.4 identified two technically feasible restoration actions for restoration of injured oyster habitats. These include the following:

- Natural Recovery; and
- Reef Restoration.

The following sections summarize available literature that documents costs for oyster reef restoration activities.

4.2.4.1 Oil Related Literature

As discussed in Chapters 2 and 3, there are no documented cases in the literature where oyster reefs were restored due to oil injury. Contacts with scientific experts and resource management personnel confirmed the absence of reef restoration efforts for oyster reef injury caused by oil discharges. As a result, there are no documented cost data for this restoration application.

4.2.4.2 Non-oil Related Literature

The associated costs of oyster reef restoration activities are documented in several sources which detail reef restoration performed as a result of structural injury to the reef habitat. The following literature sources identify costs related to reef reconstruction: Hofstetter (1981a,b); Berrigan (1988a,b, 1990); Marwitz and Bryan (1990); Bowling (1991a,b); and Soniat et al. (1991). Costs for reef reseeding using seed oysters were obtained by the Maryland Department of Natural Resources (MDNR, 1991).

4.2.4.3 Cost of Restoration Actions

4.2.4.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.4.3.2 Reef Restoration

Reef restoration includes two techniques: reconstruction of oyster reef substrate using alternative materials, and reestablishment of the habitat or other comparable site with seed oysters. The reported costs for each action are provided below.

4.2.4.3.2.1 Reef Reconstruction

Historical reef restoration projects were performed using materials suitable for reef reconstruction for oyster settlement. The placement of suitable substrate, or clutch, is a potentially successful action for increased oyster colonization if it is performed in areas with adequate bottom types (i.e., conducive for immediate oyster set) (Kennedy, 1991; Webster and Meritt, 1988). In general, oysters settle best on bottom types that are firm, such as those of rock, stone, or shell.

Cost estimates were derived from the literature for several types of materials, demonstrated as feasible actions for oyster reef restoration. These materials include shell (oyster and clam), limestone, gravel, and concrete. The reported costs of restoring injured oyster reef habitats with different materials represent the total costs for obtaining material, transportation (all phases) of material, and distribution onto the seafloor. The costs for reconstruction activities using these materials were derived based on actual restoration projects performed in specific geographical regions. Exhibit 4.8 summarizes costs for reef reconstruction for each material. Data on project costs for reef restoration are presented on a per unit basis, represented as the dollar cost per hectare of habitat restored. Costs range from approximately $3,453 to $13,896 per hectare of habitat restored and were adjusted to reflect mid-1992 dollars.

Costs for reef restoration actions vary due to different restoration requirements and commonly incorporate costs for materials, labor, and transportation requirements. The cost of materials used for reef reconstruction will vary based on the availability of useable substrate in addition to the location of the material supply. Transportation costs are generally factored into reef restoration costs due to the immediate need for material pick-up (at the supply site) and *in situ* placement (using both land and water transportation sources).

As referenced in the literature for past restoration projects, the agency sponsoring the restoration work generally performs the post-restoration monitoring activities. Cost data have historically not been provided for reef monitoring. It is assumed that reef monitoring takes place as part of routine resource management activities. Information from past reef restoration projects suggests that these costs are not generally separated in the computation of total costs for a restoration project. It is common practice for a restoration project to be performed on a contractual basis, where the costs of each activity are included in the total bid price of the restoration contract (i.e., cost of materials, transportation, and labor).

4.2.4.3.2.2 Reseeding of Mollusc Reefs

It is common practice for managers of regional oyster fisheries to cultivate seed oyster grounds for stocking purposes. Seed oysters are small, not-fully-developed oysters that are commonly raised in hatcheries or specially designated oyster beds. The rate of oyster reef restoration may be enhanced by transplanting seed oysters onto the reef site or to an established off-site reef habitat.

In a review of oyster restoration literature that identified several reef reconstruction projects performed in the past, no projects were identified that specifically performed reef restoration by reseeding the oyster bed with seed stock obtained from oyster hatcheries or private seed beds. Cost estimates of this procedure, therefore, were gathered from habitat management personnel at MDNR (MDNR, 1992). The MDNR performs annual seeding activities in the oyster beds located in the Chesapeake Bay and often restores the oyster stock in managed beds using seed material. Through information and data from management officials, unit costs for reseeding were estimated to range from approximately $1,153 to $1,339 per hectare of oyster bed reseeded (see Exhibit 4.8). These costs were derived from data on total fiscal expenditures for seed oysters and the total number of acres planted, adjusted to reflect costs in mid-1992 dollars.

The supply of oyster seed stock used for reseeding public oyster grounds is sometimes provided by privately-owned seed oyster harvesters who contract with habitat management authorities (MDNR, 1992). These suppliers commonly offer seed oysters at discount for state management purposes. Therefore, the prices at which state management agencies receive seed supply may not reflect their true market cost.

Exhibit 4.8 Reported costs for mollusc (oyster) reef restoration ($/ha in mid-1992 dollars).

Restoration Action and Material Used	Reported Cost	Source
Reef Construction		
Shell: Dredged Oyster	$7,543 - $13,896	Maryland Dept. of Natural Resources, 1992
	$3,453	Hofstetter, 1981
	$5,750	Bowling, 1991
Fresh Oyster	$1,860 - $4,323	Maryland Dept. of Natural Resources, 1992
Dredged Clamshell	$5,313	Soniat, et. al., 1991
	$12,377	Berrigan, 1988
	$10,171	Berrigan, 1990
	$3,302	Marwitz and Bryan, 1990
Limestone	$7,249	Soniat, et. al., 1991
Gravel	$6,876	Soniat, et. al., 1991
Concrete	$5,958	Soniat, et. al., 1991
Reef Reseeding		
Seed Oysters	$1,153 - $1,339	Maryland Dept. of Natural Resources, 1992

4.2.5 Coral Reef

As discussed in Section 2.2.5, the available actions for restoration of injured coral reefs include the following:

- Natural Recovery; and

- Reef Restoration using Coral Transplants.

The following sections summarize available literature on restoration costs for each action.

4.2.5.1 Oil Related Literature

The literature on restoration of coral reefs injured by oil does not identify any demonstrated actions other than allowing natural recovery to occur and monitoring. However, the use of coral transplants as a restoration method was a recommended action (Fucik et al., 1984). This method was not demonstrated in an oil discharge restoration effort and no costs for this application were documented.

4.2.5.2 Non-oil Related Literature

Restoration cost information related to coral reef restoration was documented in one report (NOAA, 1991). A recent restoration project was performed on injured corals in the Key Largo Marine Sanctuary due to a ship grounding in 1989. Associated costs of this restoration action were obtained from the National Oceanic and Atmospheric Administration (NOAA), the sponsoring agency of the restoration effort (NOAA, 1991).

4.2.5.3 Costs of Restoration Actions

4.2.5.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.5.3.2 Reef Restoration using Coral Transplants

The estimated unit costs derived for coral reef restoration were based on the restoration activities and associated costs detailed for the *M/V Elpis* grounding (NOAA, 1991). This event occurred in the Key Largo National Marine Sanctuary in November 1989. For this incident, restoration costs were estimated as part of the damage assessment.

Costs for reef restoration include the cost of activities and resources necessary to complete coral colony transplants. The unit costs estimated for restoration of coral reef habitats was based on the number of square meters of injured reef on which corals were transplanted. This unit measure of reef area is a common measure often used throughout the scientific literature to describe coral growth and cover. Cost estimates of coral reef restoration using coral colony transplants were derived from available data on the *M/V Elpis* restoration project, where several hundred square meters of reef were restored through the use of coral transplants. The cost components for restoration of this nature include the costs for labor (i.e., divers and a material handler) and materials (i.e., boat, air tanks, and supplies). These costs are summarized in Exhibit 4.9. Costs for reef restoration, adjusted to reflect mid-1992 dollars, total $236.83 per square meter of reef restored.

4.2.6 Estuarine and Marine Intertidal

This section presents estimates of costs for the restoration of estuarine and marine habitats using the action described in Section 2.2.6. Cost estimates are derived as the cost of actual restoration efforts reported in the literature, or as "engineered" estimates costing out using the techniques described in the literature. For each habitat, a range of costs is presented. Costs are also presented as unit costs (per square meter of surface area) in mid-1992 dollars. The habitats covered in this section include rocky shores, cobble-gravel beaches, sand beaches, and mud flats, as well as four bottom types.

4.2.6.1 Rocky Shore

Section 2.2.6.1 presents a discussion of the restoration actions that are relevant and feasible for this habitat. This section presents costs estimates for the following actions:

- Natural Recovery;
- Sand Blasting;
- Steam Cleaning;
- Flushing; and
- Bioremediation.

Exhibit 4.9 Reported costs for coral reef restoration ($\$/m^2$ in mid-1992 dollars).

Cost Components		Restoration Action
		Coral Colony Transplants
Labor:		
Divers (2):	Base Pay	$67.27
	Dive Pay	16.82
	Overhead/Benefits	36.66
Material Handler:	Base Pay	20.27
	Overhead/Benefits	11.05
Materials/Equipment:		
Diving Boat		80.01
Air Tanks		3.50
Cement		0.80
Plaster		0.45
Total		$236.83

Source: National Oceanic and Atmospheric Administration, 1991.

4.2.6.1.1 Oil Related Literature

Several literature sources were used and a number of experts were contacted in developing the cost estimates for rocky intertidal restoration. Moller et al. (1987) discuss steam cleaning costs. Anderson et al. (1983), Christian (1991), Hogan (1991), and R.S. Means (1990) were used to develop flushing cost estimates. In addition, Dick Lessard of Exxon was contacted for information regarding the use of chemical restoration in flushing efforts. Several literature sources were used in the development of bioremediation cost estimates, including Chianelli et al. (1991), Pritchard and Costa (1991), and Jones and Greenfield (1991). Russ Chianelli and James Bragg of Exxon, Alain Drexler and Paul Benn of Elf-Aquitaine, and Tom Merski of the National Environmental Technology Applications Corporation were also interviewed about bioremediation.

4.2.6.1.2 Non-oil Related Literature

Sand blasting costs were adapted from the Means Construction Cost Guide (R.S. Means Company, 1986). Fertilizing costs from the McMahon Heavy Construction Cost Guide (1990) were used to estimate level of effort in bioremediation.

4.2.6.1.3 Cost of Restoration Actions

4.2.6.1.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4

4.2.6.1.3.2 Sand Blasting

Cost estimates for sand blasting in rocky intertidal habitats are developed from the costs presented in the Means construction cost guide (R.S. Means Company, 1986). According to this guide, the average cost (in 1992 dollars) of sand blasting using a wet system is \$15.69 per m^2. Adding a 50 percent premium to this figure to reflect the logistics of travelling to and working in possibly remote sites or coastal areas, this unit cost is estimated to be \$23.54 per m^2.

4.2.6.1.3.3 Steam Cleaning

Moller et al. (1987) compare the costs of various steam cleaning techniques. They provide data on actual response and efforts conducted worldwide. Steam cleaning was used to restore the shoreline stained following the discharge of 22,000 barrels of heavy crude oil in the Far East. Moller et al. report the unit cost to steam clean the shoreline to be approximately \$4.82 per m^2 in this case.

4.2.6.1.3.4 Flushing

Unit cost estimates for the *in situ* treatment of rocky intertidal shores were derived assuming that washing of the shoreline is performed using a low- to medium-pressure flushing action since the action is essentially the same in any habitat. The efficiency of rock washing was estimated using data from (Anderson et al., 1983). This account of the late stages of an actual oil discharge response using flushing demonstrated that 20 to 50 m^2 of shoreline were cleaned per hour by each cleanup crew. The midpoint rate of 35 m^2 per hour was assumed as the expected efficiency of cleaning. Following discussions with several oil discharge response cooperatives and discharge response companies, unit costs were derived based on prevailing labor rates and rental rates for the necessary equipment.

A three person cleanup crew was determined as the basic unit of efficiency. Labor costs were estimated using the labor rate of the Corpus Christi Area Oil Spill Control Association (Christian, 1991), Clean Harbors (Hogan, 1991), and the Means Building Construction Cost Data (R.S. Means Company, 1990). An average per person hourly rate of $30.16 was estimated for a total labor cost of $90.47 per hour.

Equipment needs for flushing were determined to include a pressure spray unit plus a vehicle (either a truck or small boat, depending on access to the contaminated habitat). Again using figures from the Corpus Christi Area Oil Spill Control Association (Christian, 1991), Clean Harbors (Hogan, 1991), and the R.S. Means Company (1991), an approximate cost range of $30 to $40 per hour was estimated.

The cost of equipment necessary to recover and absorb dislodged contaminants were estimated using the same sources. The equipment needed for this procedure includes a small portable skimmer unit along with sorbent sweeps or booms. A cost range of $50 to $70 per hour was estimated for the recovery equipment.

The estimated total costs (including equipment and labor) for rocky intertidal shore washing and recovery in current year dollars ranged from $172.31 to $192.77 per hour. Taking the midpoint of this range as the overall estimate, a total of $182.54 per hour was estimated. Assuming the 35 m^2 per hour cleaning rate noted above, the estimated unit cost for the *in situ* treatment of rocky intertidal shore is $5.22 per m^2.

Chemical treatment of the contaminated shoreline was assumed to use a chemical such as Exxon's Corexit 9580. Corexit is sprayed onto the shoreline 15 to 30 minutes prior to the flushing operation and applied at a concentration of 0.5 to one gallon per 9.3 m^2 (i.e., per 100 ft2, Lassard, 1992). Exxon sells Corexit 9580 for approximately $16 per gallon, so at the above concentration, chemical treatment prior to flushing would add $1.30 per m^2. This yields a total for chemical treatment and flushing of $6.52 per m^2.

The following summarizes the estimated unit costs for flushing in rocky intertidal habitats:

Description	Cost per square meter
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.6.1.3.5 Bioremediation

Merski (1992) describes the application of the nutrient and microbe combination Alpha BioSea in two field situations. The first was the application of Alpha BioSea to a 12 to 16 hectare surface area of the Gulf of Mexico following the *Mega Borg* tanker discharge and fire. While this was not conducted in the intertidal zone, costs for a one-time application of the agent was approximately $38,000, or about $0.27/m^2$ of sea surface area. This same agent was used following the *Apex Barge* discharge in Galveston Bay. Following oiling of nearby marsh, Alpha BioSea was applied to approximately 30 hectares of marsh land at an approximate cost of $39,500. This equates to approximately $0.13/m^2$ of surface area. Bioremediation treatment costs in these types of efforts are likely to be much lower than in the intertidal shoreline. It should be noted that these applications of bioremediation did not produce the desired effect and that additional applications may have been necessary. Thus, this cost may represent a lower bound for a one-time application, not a comprehensive bioremediation plan.

In order to estimate costs for the application of an oleophilic nutrient to an oil-contaminated shoreline, the methods described by Pritchard and Costa (1991) and Chianelli et al. (1987) for the *Exxon Valdez* discharge cleanup are used. The basic level of operations assumes a boat is required for access to the shoreline, that tanks are used for holding and heating the nutrient, and that a crew of workers are deployed on the shoreline with backpack sprayers or hoses to apply the fertilizer.

A three person crew was determined as the basic unit of efficiency. Labor costs were estimated using the labor rate of the Corpus Christi Area Oil Spill Control Association (personal communication, 1991), Clean Harbors (Hogan, 1991), and the Means Building Construction Cost Data (R.S. Means Company, 1990). An average per person hourly rate of $30.16 was estimated, for a total labor cost of $90.47 per hour. Equipment needs for flushing were determined to include a backpack sprayer or tanks and hoses, plus a vehicle typically a small boat. Again using figures from the Corpus Christi Area Oil Spill Control Association (1991), Clean Harbors (Hogan, 1991), and the R.S. Means Company (1991), an equipment cost of $51.08 per hour was estimated. A total cost of $141.55 per hour was thus estimated for equipment and labor.

Costs for the oleophilic liquid fertilizer Inipol EAP 22 were obtained from Elf-Aquitaine, the manufacturer of the product (Benn, 1992). The following is a schedule of material costs for 440-pound drums, reflecting discounts for increased volumes:

1-9 drums	\$3.20/lb
10-24 drums	\$3.10/lb
25-79 drums	\$2.90/lb
80+ drums	\$2.65/lb

High demand for this product may necessitate additional production runs of Inipol which may drive up the cost. Bioremediation will likely be used only in light oiling situations, for small discharges, or for "polishing" operations following other response operations. Thus, it is not expected that any one bioremediation restoration effort will have the effect of driving up the material costs of Inipol.

The results from Chianelli et al. (1991) and Pritchard and Costa (1991) indicate that approximately 0.09 gallons of Inipol should be applied per m^2 of surface area. One 440-pound drum of Inipol costs \$1,408, or \$25.60 per gallon (\$1,408/55). The cost for fertilizer is thus \$2.30 per m^2 of shoreline treated.

To estimate the rate of application, figures were taken from the McMahon Heavy Construction Cost Guide (1990) for the rate of fertilizing operations. This is a reasonable surrogate for bioremediation since the basic operations are similar. According to McMahon, it requires five minutes of effort to fertilize 1,000 ft2 of a flower bed, and 30 minutes to fertilize 1,000 ft^2 of trees. These two operations are selected to represent a range of effort required depending on the terrain treated. A flower bed is similar to an easily-accessible, flat shoreline, and trees present barriers to simple spraying operations. These two operations were converted to square meters, resulting in estimated application rates of 0.0538 minutes per m^2 for rapid fertilizer deployment, and 0.3228 minutes per m^2 for slower deployment.

Using these rates, the range of cost for labor and equipment is \$0.13 to \$0.76 per m^2. Adding fertilizer costs increases that the range for total cost are \$2.43 to \$3.06 per m^2.

Material costs for the granular nutrient Customblen were not available at the time of this writing, and thus the costs for the deployment of this nutrient cannot be estimated. The cost for the spreading granular fertilizer will not likely differ greatly from the application of liquid fertilizer.

The estimation of the application of bacterial agents in the intertidal zone will also not differ from the application of nutrients only since the basic operations will likely be similar. When bacterial agents are spread on a shoreline, the operations will resemble the *Exxon Valdez* bioremediation efforts more than the application of BioSea following the *Apex Barge* discharge.

To estimate the cost for the application of bacterial agents, the same methodology used for Inipol cost estimation was utilized. Costs and volume application figures were substituted. Merski (1992) estimates costs for microbial agents to range from $20 to $25 per pound. Jones and Greenfield (1991) note that bacterial bioremediation agents cost about $15 per pound. The midpoint of these figures yields a cost of approximately $20 per pound. The rate of application of granular fertilizer following the *Exxon Valdez* discharge described by Pritchard and Costa (1991) and Chianelli et al. (1991) averaged about 0.14 pounds per m^2. Material costs are thus $2.70 per m^2 for bacterial agents. Using the same equipment and labor costs detailed above, the range of total bioremediation costs for the application of bacterial agents is $2.83 to $3.46 per m^2.

An intensive effort involving both bacteria and fertilizer over a 194-day period following a discharge of fuel oil at a power plant is described by Jones and Greenfield (1991). Efforts included site alterations to control drainage, application of nutrients, water, and bacteria, and sediment tilling to increase aeration. An area of 4,039 m^2 (44,000 ft2) was treated at a cost of $44 per metric ton. Jones and Greenfield estimate a typical range of cost for bioremediation to be $22 to $44 per metric ton, and the cost of bacteria to be approximately $33 per kilogram.

In order to estimate this unit cost per unit of surface area, the cost per metric ton figure needs to be converted. To accomplish this, an average weight per volume of twelve was derived from the McMahon Heavy Construction Cost Guide (1990). Clay, earth, mud, sand, and gravel of different degrees of compactness and moisture weigh 1.57 metric tons per m^3. Jones and Greenfield's bioremediation costs are thus evaluated on a volumetric basis, and yield unit costs of $34.60 per m^3 to $69.20 per m^3. Furthermore, Jones and Greenfield note that the oil contamination in this discharge penetrated to a depth of 15 to 20 cm. Assuming bioremediation efforts are conducted to the depth of 20 cm, and adjusting the above costs for inflation, yields a range of costs of $7.19 per m^2 to $14.39 per m^2. Note that these costs are estimated for an intensive bioremediation effort conducted on land (not the intertidal zone), and one which utilized bacterial agents. At this time, it is unclear whether adding bacterial agents contributes any additional restoration benefit in the intertidal zone since many of the earth's marine waters are rich in such agents (Chianelli, 1992).

The following table summarizes the estimated unit costs for bioremediation:

Description	Cost range per square meter
Spray Alpha BioSea from boat onto open sea	$0.27
Spray Alpha BioSea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

Among the bioremediation efforts detailed in the above table, the spraying of fertilizer or bacterial agents are among most frequently used for the rocky intertidal habitat. The first two actions in the table involve different types of habitats, while the last effort described involved agitation of sediment with disc harrows to expose soils to air. Tilling sediments is obviously not appropriate in most rocky habitats. Finally, note that the cost to add bacterial agents is not significantly different than the cost of adding nutrients.

4.2.6.2 Cobble-Gravel Beach

This section presents cost estimates relevant to the cobble-gravel intertidal habitat for the following restoration actions:

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation; and
- Bioremediation.

Cost estimates are derived from descriptions in the literature of the costs of actual field experience and engineered costs derived using techniques described in the literature.

4.2.6.2.1 Oil Related Literature

This section details the literature sources used and experts contacted in developing the cost estimates for cobble-gravel intertidal restoration. Flushing costs were estimated using Anderson et al. (1983), Christian (1991), Hogan (1991), and R.S. Means (1990). In addition, Dick Lessard of Exxon was contacted for information regarding the use of chemical restoration in flushing efforts. Sediment washing estimates were derived using Gumtz (1972), Bocard et al. (1987), Huet et al. (1989), Morris et al. (1982), Jahns et al. (1991), Michel et al. (1991), and Gundlach et al. (1991). Cost estimates for sediment agitation were developed using the American Petroleum Institute (1991), Levine (1987), Miller (1987), the U.S. EPA (1990), and the New Pig Corporation (1992). Interviews with Christian (1991), Hogan (1991), and Levine (1992) were also used. The literature sources were used in the development of bioremediation cost estimates, include Chianelli et al. (1991), Pritchard and Costa (1991), and Jones and Greenfield (1991). Russ Chianelli and James Bragg of Exxon, Alain Drexler and Paul Benn of Elf-Aquitaine, and Tom Merski of the National Environmental Technology Applications Corporation were also contacted for bioremediation information.

4.2.6.2.2 Non-oil Related Literature

Labor costs were estimated using data from the Means Construction Cost Guide (R.S. Means Company, 1990). The McMahon Heavy Construction Cost Guide (1990) was also used to estimate level of effort in bioremediation fertilizing costs.

4.2.6.2.3 Cost of Restoration Actions

4.2.6.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.6.2.3.2 Flushing

Unit cost estimates for the *in situ* treatment of cobble-gravel intertidal shores were derived assuming the same restoration actions as discussed for rocky intertidal shorelines above (see Section 4.2.6.1.3.4.). A range of costs is developed that assumes either the basic flushing operations or a chemical pre-soak with a surface washing agent followed by flushing. These operations are discussed in detail in Section 2.2.6.1.

The following summarizes the estimated unit costs for flushing in cobble-gravel intertidal habitats:

Description	Cost per square meter
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.6.2.3.3 Sediment Washing

The costs for the selected restoration actions were estimated using the results of Gumtz (1972), Bocard et al. (1987), Huet et al. (1989), and Morris et al. (1982).

Gumtz (1972) describes the costs related to development and field testing of a mobile sediment washing device constructed for the U.S. Environmental Protection Agency. The cost to construct the mobile cleaner in 1992 dollars was $249,900. Assuming this equipment still exists and is still available for restoration efforts, using such equipment in restoration efforts will not require reinvestment in equipment development. Gumtz (1972) estimates annual operation costs, including unit depreciation, labor, support functions, and maintenance, to be the equivalent of $48,800 in 1992 dollars. Gumtz estimates a unit cost for operations of about $2.49 per metric ton of sand. Using the average weight per volume for three types of sand gravel presented in McMahon (1990), an overall average of 1.74 metric tons per m^3 was estimated. The cost for operations is thus $4.34 per m^3.

In addition to the Gumtz cost estimate, an average sand cleaning cost of approximately $50 per m^3 of beach material was derived using the technology described in Bocard (1987) and Huet et al. (1989). Using the technology described in Bocard et al. (1987), an average cleaning cost of approximately $50 per m^3 of beach material was estimated (Huet et al., 1989). This figure was adjusted for inflation and to reflect the overhead and profit charges expected in response and restoration contracts. Overhead and profit was assumed to be 25 percent (R.S. Means Company, 1990). This cleaning cost was then adjusted for inflation, leading to an estimate of $69.12 per m^3 of contaminated beach material cleaned.

Morris et al. (1982) describe sediment washing in an actual case of an oil discharge restoration project. The results of this effort indicated that it cost approximately $57.58 per m^3 of beach material. Adjusting for inflation yields an estimate of $80.36 per m^3.

The midpoint of the two cost estimates above is $74.74 per m^3. Because of the variable depth to which intertidal habitats may be oiled, it is necessary to estimate costs for cobble and gravel habitats separately from sand habitats.

An overall range of estimates for using the beach washing technology for the *in situ* restoration of cobble and gravel intertidal shorelines was derived by adjusting the Gumtz estimate of \$4.34 per m^3 and the engineered cost of \$74.74 per m^3 to reflect the depth of beach material to be removed. The results Jahns et al. (1991), Michel et al. (1991), and Gundlach et al. (1991) were used in this estimation. Jahns et al. (1991) noted oil penetration of 50 to 100 cm in these environments. Michel et al. (1991) estimated penetration to occur 25 to 50 cm, and Gundlach et al. (1991) noted 40 to 60 cm depth. Taking a rough midpoint of 50 cm, it was assumed that 53 cm of gravel or cobble beach material would need to be removed for cleaning. This depth includes a 3 cm buffer to represent the imprecision of beach removal equipment or human error in removing exactly 50 cm of material. Assuming that cleaning crews dig to 53 cm, estimates for cleaning cobble and gravel intertidal habitats range from \$2.30 to \$39.61 per m^2 of beach surface area.

The range of costs appears to arise from the volume of material processed. Whereas Gumtz estimates operating costs on an annual basis (equivalent to about 11,250 m^3 of sand), Huet et al. base their estimate on the cleaning of just 1,800 m^3 of pebbles. Therefore, fixed costs, such as equipment set-up, will be higher per unit of surface area for small discharges. The range of costs presented below should thus be viewed relative to the size of a discharge event. Small discharges will likely have higher *unit* costs, and vice versa.

The following table summarizes the estimated unit costs for sediment washing in cobble-gravel intertidal habitats:

Description	Cost range per square meter
Mobile sediment washer	\$ 2.30
Engineered cost estimate	\$39.61

4.2.6.2.3.4 Sediment Agitation

Levine notes that the total cost for development and deployment of two Muck Monster boats and two Muck Monster bulldozers fell into the range of \$1.5 to \$2 million (American Petroleum Institute, 1991). Since Levine (1987) describes the area in which the restoration effort was conducted (2,134 linear meters of shoreline was cleaned to a width of about 27.5 meters), a cost range may be estimated per unit of surface area. This area equates to 58,550 m^2, yielding a unit cost range in 1992 dollars of \$26.21 to \$34.95 per m^2.

Since future shoreline agitation efforts will not be faced with the development costs to create a Muck Monster. Future agitation costs are expected to be lower. Levine (1992) stated that a similar effort conducted now could be performed at a reduced cost. Assuming that the expertise is readily available, and that no experimental efforts are undertaken during restoration efforts, the following shoreline agitation costs are estimated for the use of the Muck Monster technology. (Note that this particular technology is patented by Arco Marine, Inc., who must be contacted prior to the use of the Muck Monster.)

This estimate takes into account all items needed for shoreline agitation, and estimates their cost based on the published costs for various items. Based on Levine (1987), Miller (1987), and conversations with Levine (1992), the following equipment was used in the *Arco Anchorage* restoration: sorbent boom, sweeps, two bull dozers or log skidders, two water pumps, a vacuum truck, two skiffs, and approximately 18 personnel.

Costs for equipment were determined using price lists presented for an oil discharge cooperative (Christian, 1991), the Environmental Protection Agency (U.S. EPA, 1990), and a commercial catalog (New Pig, 1992). Whenever possible, the midpoint of a range of multiple sources was used to represent costs of different types of organizations. Labor costs were estimated using the labor rate of the Corpus Christi Area Oil Spill Control Association (Christian, 1991), Clean Harbors (Hogan, 1991), and the Means Building Construction Cost Data (R.S. Means Company, 1990). All cost are in mid-1992 dollars.

The following table presents itemized cost estimates for shoreline agitation using the Muck Monster:

Sorbent Boom	$ 27,888.00
Sweeps	$ 53,922.00
Bull Dozer/Log Skidder (2)	$102,782.40
Pumps (2)	$ 10,965.36
Vacuum Truck	$ 29,400.00
Skiffs (2)	$ 16,128.00
Labor (18)	$312,699.00
Total	$553,784.76

As noted above, the restoration following the *Arco Anchorage* discharge covered a surface area of 58,550 m^2. The expected cost of the efficient use of the Muck Monster technology is thus $9.46 per m^2.

The following table summarizes the estimated unit costs for shoreline agitation:

Description	Cost range per square meter
Efficient use of Muck Monster technology	$9.46
Costs for Muck Monster development and operation	$26.21 to $34.95

4.2.6.2.3.5 Bioremediation

Bioremediation efforts in cobble-gravel intertidal habitats are essentially the same as those described in detail in the rocky intertidal sections of this report (see Sections 2.2.6.1.3.5. and 4.2.6.1.3.5.). This effort provides a range of cost figures for various bioremediation efforts, although as the following table shows, there is not a great deal of variation in the cost of basic operations:

Description	**Cost range per square meter**
Spray Alpha BioSea from boat onto open sea	$0.27
Spray Alpha BioSea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

Realistically, the actions most likely to be used for the cobble-gravel intertidal habitat are the adding of fertilizer or bacterial agents described above. The basic addition of fertilizer to contaminated soil and the adding of bacterial agents do not differ significantly in cost, and their ranges in fact overlap.

4.2.6.3 Sand Beach

Many actions for the restoration of sandy intertidal habitats are similar to those used in cobble-gravel restoration. Differences in methods exist for this habitat, however, due to the increased ecological sensitivity and different penetration of oil into sediments for sand and gravel. Cost estimates are presented below for the following restoration methods:

- Natural Recovery;
- Flushing;
- Sediment Washing;

- Sediment Agitation;
- Bioremediation; and
- Incineration.

4.2.6.3.1 Oil Related Literature

Due to similarities in the habitats, the literature sources used and experts contacted in developing the cost estimates for sand intertidal restoration are the same as those described in Section 4.2.6.2.1. Holoboff and Foster (1987), however, were used rather than Jahns et al. (1991), Michel et al. (1991), and Gundlach et al. (1991) to estimate the depth of oil penetration in sand sediments.

4.2.6.3.2 Non-oil Related Literature

Information obtained through personal communication with Garbaciak (1992), which detailed incineration of contaminated sand sediments, was used in developing incineration cost estimates.

4.2.6.3.3 Cost of Restoration Actions

4.2.6.3.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.6.3.3.2 Flushing

Unit cost estimates for the *in situ* treatment of sand intertidal shores were derived assuming the same restoration actions as discussed for rocky intertidal shorelines above (see Section 4.2.6.1.3.4.). Christian (1991) indicates the appropriateness of this action for sand shorelines. A range of costs is developed that assumed either the basic flushing operations or a chemical pre-soak with a surface washing agent followed by flushing. These operations are discussed in detail in Section 2.2.6.1.

The following summarizes the estimated unit costs for flushing in sand intertidal habitats:

Description	Cost per square meter
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.6.3.3.3 Sediment Washing

The costs for sediment washing were adapted using the same actions assumed for cobble-gravel shorelines (see Section 4.2.6.2.3.3. above). The primary difference between the washing of sediment from cobble-gravel or sand beaches is the difference in the depth to which oil will penetrate in either environment. Holoboff and Foster (1987) report an estimated penetration of 30 to 40 cm in their experimental data. Using their 30 cm figure, it was assumed that at least 33 cm of sand would have to be removed for cleaning. This would allow for a 3 cm buffer for error in the use of any digging or bulldozing type of machinery, which are not precise enough to extract exactly 30 cm of soil. Assuming that cleaning crews will dig to a depth of 33 cm, an overall estimate for cleaning sand intertidal habitats of $24.65 per m^2 of beach surface area was derived. The lower bound of the estimated range of costs is taken from Gumtz (1972).

The following summarizes the estimated unit costs for sediment washing in sand intertidal habitats:

Description	Cost range per square meter
Field experimentation with mobile sediment washer	$ 2.30
Engineered cost estimate	$24.65

4.2.6.3.3.4 Sediment Agitation

The estimated costs for shoreline agitation in sand intertidal habitats are estimated to be the same as those for agitation in cobble-gravel environments (see Section 4.2.6.2.3.4.). Again, the range of possible costs given includes the use of Arco Marine's Muck Monster technology and development and deployment of some other type of sediment agitating technology.

The following table summarizes the estimated unit costs for shoreline agitation:

Description	Cost range per square meter
Efficient use of Muck Monster technology	$9.46
Costs for Muck Monster development and operation	$26.21 to $34.95

4.2.6.3.3.5 Bioremediation

Bioremediation efforts in sand intertidal habitats are essentially the same as those described in detail in the rocky intertidal sections of this report (Sections 2.2.6.1.3.5. and 4.2.6.1.3.5.). This effort provides a range of cost figures for various bioremediation efforts, although as the following table shows that there is not a great deal of variation in the cost of basic operations:

Description	Cost range per square meter
Spray Alpha BioSea from boat onto open sea	$0.27
Spray Alpha BioSea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

As in the case of the other intertidal habitats, the actions most likely to be used for the sand intertidal habitat include the addition of fertilizer to contaminated soil or the addition of bacterial agents. These two actions, however, do not differ significantly in cost.

4.2.6.3.3.6 Incineration

Unit costs for the incineration of sediments in sand intertidal habitats were estimated using project cost data provided by the U.S. Army Corps of Engineers (Garbaciak, 1992). Costs were provided for a number of incineration related activities. The different activities are used to estimate a range of incineration unit costs. The following table shows the costs per volume of material for a number of activities:

Description	Cost per cubic meter
Incinerate material	261.58
Remove large debris	$2.62
Dewater material	$3.92
Rehandle material into incinerator	2.62
Treat removed water	$5.23
Solidify ash	$58.85
Rehandle material into disposal site	$1.31
Total	$336.13

The cost, therefore, of an intensive incineration effort, including a number of incineration activities, is $336.12 per m^3. To develop a lower end of the range of incineration costs, the cost of incineration alone, $261.58 per m^3, was used.

In order to convert these costs to costs per surface area, the results of Holoboff and Foster (1987) described in detail for sediment washing were used. As indicated above, approximately 33 cm of sand sediment is likely to be removed for restoration. Assuming oil penetrates to a depth that requires this much material be removed, the following range of unit costs per surface area for incineration of sand sediments is estimated:

Description	Cost range per square meter
Incineration of sand alone	$ 86.32
Intensive incineration effort	$110.92

4.2.6.4 Mud Flat

This section presents a discussion of the costs of restoration actions appropriate to mudflats. Specifically, cost estimates are presented for the following:

- Natural Recovery;
- Sediment Removal and Replacement; and
- Bioremediation.

4.2.6.4.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.6.4.2 Sediment Removal/Replacement

A restoration method was assumed which involves removing contaminated soil, loading and transporting it for disposal, and obtaining and deploying replacement soil in its place. Cost estimates for removing contaminated soil, and purchasing and spreading new soil were obtained from *Means Building Construction Cost Data 1991* (R.S. Means Company, 1990). Cost data for loading, trucking, and disposing contaminated soil were gathered to represent a range of conditions and scenarios. Calculated costs were also converted to mid-1992 price levels.

Stripping, new soil, and spreading costs were calculated per m^3 of soil. A 100 percent premium was added to the Means construction cost figures to represent the additional effort required to deal with wet material expected in the intertidal zone. The costs estimated were as follows: stripping and piling on-site for loading, \$6.34 per m^3; replacement soil (screened loam), \$29.42 per m^3; and spreading new material from pile to rough finish grade, \$9.61 per m^3. This results in a total cost of \$45.37 per m^3 of contaminated soil for stripping, new soil, and replacement.

Exhibit 4.10 presents the disposal costs estimated by several sources. The organizations contacted are located throughout the United States and consist of one oil discharge response cooperative, two oil discharge response and/or remediation companies, and two hazardous waste management companies. Since the contaminant is assumed to be non-hazardous, the soil need not be treated or stabilized prior to disposal in an upland landfill. The cost estimated by Clean Harbors was per 55 gallon drum disposed; volume discounts are expected when disposal occurs with larger containers. Since the clean Harbors estimate is considerably higher than the others in Exhibit 4.10 it was treated as an outlier and omitted from the calculation of an overall unit cost for disposal. The high range of the remaining estimates were used to calculate the overall average disposal cost of \$166.82 per m^3 of soil.

Harper and Humphrey (1985) note that oil penetration in mud flats occurs to a depth range of 2 to 4 cm. Unit costs are calculated per square meter of surface area assuming that soil is contaminated to a depth of 2 cm. As in other sections, a 3 cm buffer is added to the assumed penetration depth to reflect the imprecision of digging to an exact depth. Overall, 5 cm of soil are assumed removed and disposed. As a result, the overall costs are assumed to be \$2.27 per m^2 of mud flat surface area for restoration and \$8.35 per m^2 for disposal.

The following table summarizes the estimated unit costs for removal, replacement, and disposal of contaminated mud flat intertidal sediments:

Description	Cost range per square meter
Removal and replacement	$2.27
Disposal	$8.35

Exhibit 4.10 Reported monitoring costs for coral reef restoration using coral transplants ($/m^2 in mid-1992 dollars).

Cost Component		**Duration of Monitoring**				
		Year 1	**Year 2**	**Years 3-5**	**Years 6-10**	**Total Years 1-10**
Labor						
Field:	Principal Investigator	$2.34	$1.14	$0.68	$0.45	$5.06
	Biologists	8.22	1.81	1.08	0.36	11.47
Analysis:	Principal Investigator	18.70	4.52	2.71	1.81	27.74
	Biologists	30.00	7.25	4.36	1.45	43.06
Materials						
Field:	Boat	4.87	2.35	1.41	0.94	9.57
	Supplies	0.79	0.19	0.19	0.19	1.37
Analysis:	Supplies	0.39	0.32	0.32	0.32	1.34
Total Labor and Materials		65.31	17.59	10.75	5.52	99.17
Overhead (100%)		65.31	17.59	10.75	5.52	99.17
Total Monitoring Costs		$130.62	$35.18	$21.50	$11.04	$198.34

Source: National Oceanic and Atmospheric Administration, 1991.

4.2.6.4.3 Bioremediation

Bioremediation in mud flat intertidal habitats is estimated to be the same as estimated for other intertidal habitats. The following table shows the range in the cost of basic bioremediation operations:

Description	Cost range per square meter
Spray Alpha BioSea from boat onto open sea	$0.27
Spray Alpha BioSea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

As in the case of the other intertidal habitats, the actions most likely to be used for the mud flat intertidal habitat include the addition of fertilizer or bacterial agents to contaminated soil.

4.2.7 Estuarine and Marine Subtidal

4.2.7.1 Subtidal Rock Bottom

There is only one technical feasibility restoration action applicable to subtidal estuarine and marine rock bottom habitats: natural recovery. The costs of monitoring programs are discussed in Section 4.4.

4.2.7.2 Subtidal Cobble-Gravel, Sand, and Silt-Mud Bottom

Section 2.2.7.2 discusses the technical feasibility of restoration actions for subtidal estuarine and marine cobble-gravel, sand, and silt-mud bottom habitats including:

- Natural Recovery;
- Dredging/Sediment Removal; and
- Sediment Capping.

The following sections identify available information on the costs of these actions and summarize reported cost estimates.

4.2.7.2.1 Oil Related Literature

There are no costs reported for oil related restoration of bottom sediments by material removal in cobble-gravel, sand, and silt-mud habitats. Representative cost sources for sediment removal due to other forms of contamination are identified in Section 4.2.7.2.2.

4.2.7.2.2 Non-oil Related Literature

Sediment removal and disposal costs are documented in several literature sources to reflect activities performed in various geographic regions of the United States. Also, recent data were gathered on the costs of actual dredging and disposal activities performed by the USACOE. Phillips and Malek (1987) identify costs related to alternative dredging methods for use in the Puget Sound, Washington area. These costs reflect costs of material removal only, yet present a representative average of equipment costs for the northwestern region of the U.S. Similar costs for the Puget Sound region are identified Cullinane et al. (1990). Eastern Research Group (ERG) (1991) documents a range of cost information for dredging and disposal activities for the management of contaminated sediments. These cost data are provided for sediment removal activities alone, as well as for sediment removal and disposal for three types of disposal methods: open water, near-shore confined, and upland disposal.

Marcus (1992) summarizes costs for contaminated material dredging and disposal for near-shore disposal. Cost data were also provided by the USACOE for maintenance and new work dredging operations performed in 1990 and 1991 in several USACOE districts (USACOE, 1992). The operation costs were categorized based on different types of dredge equipment used and the disposal method selected (i.e., open water, near-shore confined, upland disposal). These data reflect the unit costs of dredging and disposal activities routinely performed by the USACOE and are considered representative data for the majority of dredging projects conducted in various regions of the United States.

4.2.7.2.3 Cost of Restoration Actions

Costs of subtidal cobble-gravel, sand, and silt-mud bottom restoration activities are discussed in the following subsections.

4.2.7.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.7.2.3.2 Dredging/Sediment Removal

The costs of material removal using dredging equipment for cobble-gravel, sand, and silt-mud habitats typically include three main components: the type of dredge equipment used, transport mode, and associated disposal methods. Actual costs of dredging operations were provided by several sources (Phillips and Malek, 1987; Cullinane et al., 1990; ERG, 1991; Marcus, 1991; and USACOE,1992). Based on a review of material removal costs identified in the literature, costs are presented for just material removal activities (i.e., dredging). Costs are also provided which combine dredging, transport, and disposal activities. Each category of costs is presented in Exhibit 4.11.

Material removal costs available which provide costs only for dredging range from approximately $1.38 per cubic meter to $1.54 per cubic meter (Phillips and Malek, 1987; Cullinane et al., 1990). These costs were adjusted to reflect current dollars. This range of costs reflects differences in the type of equipment used, site specific factors, and costs of operations in a given geographic region.

Exhibit 4.11 also identifies a range of costs associated with sediment removal and the associated transport and disposal of the dredged material. These cost components are not broken out for each activity, but shown as a combined cost. Cost information identified in the literature is often specific to the job characteristics in a given region of the United States and therefore costs are not typically documented for each cost component. Costs will vary based on the sediment management strategy selected to appropriately deal with different levels of contaminated sediment, as identified by the range of dredging and disposal costs presented in Exhibit 4.11. The cost ranges presented for each disposal method represent typical costs of performing the specific method, and may vary based on factors such as the level of contamination, type of equipment selected, geographic region as well as the availability of and distance to disposal locations. To develop a reasonably accurate estimate of dredging and disposal costs, one must identify cost components region by region, if not project area by project area. According to ERG (1991), costs can vary significantly even between adjacent areas because of the factors listed above as well as differences in mobilization and demobilization costs. In addition, monitoring, enforcement, and regulatory costs will add to these disposal costs. These cost components are typically not included in documented cost information.

In cases where the costs of transport are not included with the costs of dredging (in order to move dredged sediment to offshore or upland disposal sites), transportation costs may accrue significantly depending on the size of the project. Two literature sources identified cost estimates for two transport modes: truck and barge hauling (Phillips and Malek, 1989; Cullinane et al., 1990).

Exhibit 4.11 Reported costs for subtidal estuarine and marine cobble-gravel, sand, and silt-mud bottom sediment removal and disposal ($\$/m^3$ in mid-1992 dollars).

Sediment Removal

Source	Costs	Other Data Reported
Phillips and Malek (1987);		
Cullinane, et al., (1990)		
Hydraulic Dredging		
Mechanical Dredging	$1.38	Northwestern U.S.
Hopper Dredging	$1.48	Northwestern U.S.
	$1.28	Northwestern U.S.
ERG (1991)	$1.54	Average cost U.S.

Sediment Removal and Disposal

Disposal Method	Source	Costs	Other Data Reported
Open Water	ERG (1991) USACE (1992)	$4.25 $0.32 - $10.17	Northeastern U.S. All U.S. Regions
Near-Shore Confined	ERG (1991) Marcus (1991) USACE (1992)	$4.75 - $11.41 $0.30 - $8.97 $0.22 - $20.20	Northeastern U.S. Great Lakes Regions All U.S. Regions
Upland	ERG (1991) USACE (1992)	$4.75 - $11.21 $0.36 - $11.58	Northeastern U.S. All U.S. Regions

Each source reported similar costs, as outlined below:

Source	Cost	Dredged Material Transportation Costs ($/m^3/mile)
Phillips and Malek (1987)	$0.25	Truck transport
Cullinane et al. (1990)	$0.25 -$0.30	Barge transport

4.2.7.2.3.3 Sediment Capping

The costs of capping sediments reflect the costs of obtaining clean dredged material or some other form of loam, sand, or fill which is dredged locally (or taken from a maintenance dredging operation), transported to the disposal, and placed on the contaminated sediment. The capping material is assumed to be dredged at a similar unit cost. Typically a ratio of four parts of capping material to one part contaminated dredge material is considered an appropriate amount of material needed for contaminant isolation (Cullinane et al., 1990; USACOE, 1989; Averett and Palermo, 1989). Available costs of capping material were documented in the literature (Phillips and Malek, 1987; Cullinane et al., 1990; and ERG, 1991). These sources identify costs for capping material to range from about $1.22 per cubic meter to $4.25 per cubic meter. The following table provides a breakout of these cost estimates

Sediment Capping Costs ($/m^3)

Source	Cost	Other Data Reported
Phillips and Malek (1987)	$1.29	Northwestern U.S
Cullinane et al., (1990)	$1.29	Northwestern U.S
ERG (1991)	$4.25	Northeastern U.S

4.2.8 Riverine and Lacustrine Shorelines

The following discusses the costs of the restoration actions considered for riverine and lacustrine (lake) shoreline habitats.

4.2.8.1 Rock Shore

- Natural Recovery;
- Sandblasting;
- Steam Cleaning;

- Flushing; and

- Bioremediation.

4.2.8.1.1 Oil Related Literature

The oil related literature used to develop cost estimates for restoration actions for riverine and lacustrine rocky shore environments is the same as that presented in Section 4.2.6.1.

4.2.8.1.2 Non-oil related Literature

The non-oil literature used to develop cost estimates for restoration actions for riverine and lacustrine rocky shore environments is the same as that presented in Section 4.2.6.1.

4.2.8.1.3 Cost of Restoration Actions

4.2.8.1.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.8.1.3.2 Sandblasting

Cost estimates for sand blasting in riverine and lacustrine rocky shore habitats are developed in Section 4.2.6.1.3.2. A cost range of \$15.69 to \$23.54 per m^2 was estimated for sand blasting rocky shores.

4.2.8.1.3.3 Steam Cleaning

Using Moller et al. (1987), Section 4.2.6.1.3.3. estimated the unit cost to steam clean rocky shoreline at approximately \$4.82 per m^2. This cost would apply to riverine and lacustrine environments as well.

4.2.8.1.3.4 Flushing

The cost estimates derived in Section 4.2.6.1.3.4. for rocky intertidal habitats also applies to riverine and lacustrine rocky shore environments. The following summarizes the estimated unit costs for flushing in rocky shore habitats:

Description	Cost per square meter
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.8.1.3.5 Bioremediation

The following table summarizes the unit costs estimated in Section 4.2.6.1.3.5. for bioremediation. These costs apply to riverine and lacustrine environments:

Description	Cost Range per square meter
Spray Alpha Bio-Sea from boat onto open sea	$0.27
Spray Alpha Bio-Sea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

4.2.8.2 Cobble-Gravel Shore

The costs derived for intertidal cobble-gravel shorelines apply to riverine and lacustrine environments as well. The costs derived for the following actions are summarized below.

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation; and
- Bioremediation.

4.2.8.2.1 Oil Related Literature

See Section 4.2.8.2. for the oil related literature used to estimate cobble-gravel shore restoration.

4.2.8.2.2 Non-oil Related Literature

See Section 4.2.8.2. for the non-oil literature used to estimate cobble-gravel shore restoration.

4.2.8.2.3 Cost of Restoration Actions

4.2.8.2.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.2.8.2.3.2 Flushing

Section 4.2.6.1.3.4. provides detailed derivations of unit costs for flushing in cobble-gravel shore habitats. The following summarizes the estimated unit costs:

Description	**Cost per square meter**
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.8.2.3.3 Sediment Washing

The following table summarizes the estimated unit costs derived in Section 4.2.6.2.3.3. for sediment washing in cobble-gravel shore habitats:

Description	**Cost per square meter**
Mobile sediment washer	$2.30
Engineered Cost Estimate	$39.61

4.2.8.2.3.4 Sediment Agitation

The following table summarizes the estimated unit costs for sediment agitation. See Section 4.2.6.2.3.4. for a full derivation of these costs.

Description	Cost Range per square meter
Efficient use of Muck Monster technology	$9.46
Costs for Muck Monster development and operation	$26.21 to $34.95

4.2.8.2.3.5 Bioremediation

The following range of cost figures for various bioremediation efforts was developed in Section 4.2.6.1.3.5:

Description	Cost Range per square meter
Spray Alpha Bio-Sea from boat onto open sea	$0.27
Spray Alpha Bio-Sea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

Practically, the actions most likely used for the cobble-gravel shore habitat include the use of bacterial agents or intensive terrestrial oil removal efforts with bacterial agents described above. The basic addition of fertilizer to contaminated soil and adding of bacterial agents do not differ significantly in cost and their ranges in fact overlap.

4.2.8.3 Sand Shore

- Natural Recovery;
- Flushing;
- Sediment Washing;
- Sediment Agitation;
- Bioremediation; and

- Incineration.

4.2.8.3.1 Oil Related Literature

The oil related literature used to develop cost estimates for restoration in riverine and lacustrine sand shore environments is the same as that used for sand intertidal habitats (see Section 4.2.6.3.).

4.2.8.3.2 Non-oil Related Literature

See Section 4.2.6.3. for a discussion of the non-oil literature used to develop restoration cost estimates.

4.2.8.3.3 Cost of Restoration Actions

The cost of restoration activities in riverine and lacustrine shorelines is the same as the cost for intertidal sand beach restoration.

4.2.8.3.3.1 Natural Recovery

Section 4.4. provides a description of the costs of monitoring programs.

4.2.8.3.3.2 Flushing

The following summarizes the estimated unit costs for flushing in sand shore habitats. These costs are detailed in Section 4.2.6.1.3.4:

Description	**Cost per square meter**
Flushing	$5.22
Flushing with chemical treatment	$6.52

4.2.8.3.3.3 Sediment Washing

The following table summarizes the estimated unit costs for sediment washing in sand shore habitats. These costs are described in detail in Section 4.2.6.2.3.3.

Description	**Cost per square meter**
Field experimentation with mobile sediment washer	$ 2.30
Engineered cost estimate	$24.65

4.2.8.3.3.4 Sediment Agitation

The following table summarizes the estimated unit costs for shoreline agitation that were derived in detail in Section 4.2.6.2.3.4.:

Description	Cost Range per square meter
Efficient use of Muck Monster technology	$9.46
Costs for Muck Monster development and operation	$26.21 to $34.95

4.2.8.3.3.5 Bioremediation

The following range of cost figures for various bioremediation efforts was developed in Section 4.2.6.1.3.5.:

Description	Cost Range per square meter
Spray Alpha Bio-Sea from boat onto open sea	$0.27
Spray Alpha Bio-Sea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

As in the case of the other freshwater shorelines, the actions most likely used for the sand shore habitat include the addition of fertilizer to contaminated soil or adding of bacterial agents. These two actions, however, do not differ significantly in cost.

4.2.8.3.3.6 Incineration

Unit costs for the incineration of sediments in sand shore habitats were estimated in Section 4.2.6.3.3.6. The following table shows a range of unit costs for incineration of sand shore sediments:

Description	Cost Range per square meter
Incineration of sand alone	$86.32
Intensive incineration effort	$110.92

4.2.8.4 Silt-Mud Shore

The actions costed for this habitat were as follows:

- Natural Recovery;
- Sediment Removal/Replacement; and
- Bioremediation.

4.2.8.4.1 Oil Related Literature

See Section 4.2.6.4. for information on the oil related literature used to derive cost estimates for mud flat restoration. In addition to the sources used for mud flat restoration, the American Petroleum Institute (1991) provided an engineered cost estimate for sediment removal and replacement.

4.2.8.4.2 Non-oil Related Literature

See Section 4.2.6.4. for the non-oil literature used in the development of restoration cost estimates.

4.2.8.4.3 Cost of Restoration Actions

In general, the costs estimated for mud flat intertidal habitats also applies to silt-mud riverine and lacustrine shorelines. An additional estimate is provided for sediment removal and replacement, however, for the silt-mud riverine habitat.

4.2.8.4.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.2.8.4.3.2 Sediment Removal/Replacement

In addition to the costs derived in Section 4.2.6.4.3.2., the American Petroleum Institute (1991) presents cost estimates for a discharge scenario in which contaminated streambank sediments are removed and replaced. In order to remove and replace 100 cubic yards of soil, the API estimates 48 hours of labor (at $35 per hour), $2,000 for equipment, and 60 percent of labor and equipment for the disposal or treatment of soil, procurement of replacement soil, and contingency. These cost items total $7,701 for 100 square meters of soil, or $77.01 per m^2. Digging to the five centimeter depth assumed in Section 4.2.6.4.3.2, this equates to a unit cost of $3.85 per m^2 of silt-mud, or $3.94 per m^2 in 1992 dollars. The following table summarizes the estimated unit costs for removal, replacement, and disposal of contaminated silt or mud shore sediments. This table includes the costs estimated in Section 4.2.6.4.3.2 for removal, replacement, and disposal of mud flat sediments.

Description	**Cost per square meter**
Streambank restoration (removal, replacement, and disposal)	$3.94
Removal and replacement	$2.27
Disposal	$8.35

4.2.8.4.3.3 Bioremediation

Bioremediation in silt or mud habitats is estimated to be the same as estimated for other habitats. The following table summarizes the range in the cost of basic bioremediation operations:

Description	**Cost Range per square meter**
Spray Alpha Bio-Sea from boat onto open sea	$0.27
Spray Alpha Bio-Sea from boat onto mangroves	$0.13
Add nutrients only	$2.43 to $3.06
Add bacterial agents	$2.83 to $3.46
Intensive terrestrial effort with bacterial agents	$7.19 to $14.39

As in the case of the other habitats, the actions most likely to be used for the mud flat habitat include the addition of fertilizer to contaminated soil or adding of bacterial agents. See Section 4.2.6.1.3.4. for the full derivation of this range of costs.

4.2.9 Riverine Bottom

4.2.9.1 Rock Bottom

There is only one technically feasible restoration action applicable to riverine rock bottom habitats: natural recovery. Section 4.4 provides a description of the costs of monitoring programs.

4.2.9.2 Cobble-Gravel, Sand, and Silt-Mud Bottom

Section 2.2.9.2. discusses the technical feasibility of restoration actions for riverine cobble-gravel, sand, and silt-mud bottom habitats. These actions include:

- Natural Recovery;
- Dredging/Sediment Removal; and
- Streambed Agitation.

The following sections identify available information on the costs of these actions and summarize reported cost estimates.

4.2.9.2.1 Oil Related Literature

The only documented case of costs for oil related restoration performed in riverine habitats is provided as a case study on restoration in high-energy river and stream habitats (API, 1991). This report estimates costs related to a fuel discharge restoration project performed in Wolf Lodge Creek, Idaho, as documented by Graves (1985). Sediment agitation was performed as the primary restoration action. Costs of performing this action were derived from assumptions about the level of effort required for each activity and cost of equipment and supplies (API, 1991).

4.2.9.2.2 Non-oil Related Literature

No literature is available that documents actual costs of non-oil related restoration performed in riverine habitats. Refer to Section 4.2.7.2.2 for a review of literature that identifies sediment removal costs. The cost information identified for estuarine and marine subtidal bottom habitats is applicable to similar activities performed in riverine habitats.

4.2.9.2.3 Cost of Restoration Actions

Costs of monitoring programs are discussed in Section 4.4.

4.2.9.2.3.1 Natural Recovery

There are no reported costs for restoration actions related to natural recovery.

4.2.9.2.3.2 Dredging/Sediment Removal

Costs of this restoration option are discussed in Section 4.2.7.2.3.2.

4.2.9.2.3.3 Streambed Agitation

The actual costs of streambed agitation, as performed in the Wolf Lodge Creek discharge (Graves, 1985) are not reported by any documented source. However, the costs of performing the sediment agitation action as an option to restore oiled streambeds were estimated based on typical manpower, equipment, and materials requirements. These costs were derived, as reported in API (1991), based on an estimate of the number of hours required to operate equipment as well as costs of using the equipment and other materials. To reflect restoration costs on an areal basis, the derived estimates were converted to unit costs and adjusted to current dollars, as shown below:

Sediment Agitation

Cost Components	Unit Cost ($/m^2)
Labor	$0.02
Equipment	$0.01
Total	$0.03

4.2.10 Lacustrine Bottom

4.2.10.1 Rock Bottom

Section 2.2.10.1. discusses the technical feasibility of one restoration option applicable to lacustrine rock bottom habitats: natural recovery. There are no documented costs in the literature related to this action for lacustrine rock bottom habitats.

4.2.10.2 Cobble-Gravel, Sand, and Silt-Mud Bottom

Section 2.2.10.2. discusses the technical feasibility of restoration actions for lacustrine cobble-gravel, sand, and silt-mud bottom habitats. These actions include:

- Natural Recovery;
- Dredging/Sediment Removal; and
- Sediment Capping.

The following sections identify available information on the costs of these actions and summarize reported cost estimates.

4.2.10.2.1 Oil Related Literature

For subtidal lake restoration, there are no documented costs associated with restoration actions performed due to oil contamination.

4.2.10.2.2 Non-oil Related Literature

Actual cost information related to lake restoration activities is documented in a handful of sources that identify sediment removal as a common lacustrine restoration action. Costs for other restoration actions specific to lake restoration were not found in the literature (e.g., *in situ* sediment capping). Sediment removal costs for dredging activities performed in lake habitats are identified by Peterson (1978, 1982), Cooke (1983), EPA (1988), and Averett et al. (1990). Although there is a significant span of time covered by these documented cases, the cost ranges identified, after adjustment to current dollars, are reasonable estimates relative to site-specific factors.

4.2.10.2.3 Cost of Restoration Actions

Costs of subtidal riverine bottom restoration actions are discussed in the following subsections.

4.2.10.2.3.1 Natural Recovery

Costs of monitoring programs are discussed in Section 4.4.

4.2.10.2.3.2 Dredging/Sediment Removal

Actual costs of sediment removal performed in lakes are reported by Peterson (1978, 1982), Cooke (1983), EPA (1988), and Averett et al. (1990). As discussed in Section 4.2.7.2.3.2, the costs of dredging sediment are affected by several factors. These can include the rate of dredging, quantity of material removed, availability of equipment, operational constraints, and other site specific factors. Exhibit 4.12 provides a summary of costs for lake sediment removal as documented by the above mentioned literature sources. Peterson (1978, 1982) summarizes costs for several lake restoration projects and points out the difficulty in costing out dredging projects on a comparative basis. He also describes why cost figures for freshwater lake dredging projects are less common than are for USACOE navigation projects, citing that until recently there were no federal funds available to conduct or monitor projects. The cost range reported for various lake dredging projects differ due to site specific factors, equipment used, and geographical constraints. Based on available cost data for different regions where lake sediment removal was conducted, it appears that restoration of lakes in the Northeast is much more costly that for other parts of the country. Peterson (1982) also states that removal of contaminated material may increase the costs by three to five times.

Cooke (1983) provides cost estimates for lake sediment removal which range from a few dollars a cubic meter to over $30.00 for removal and handling of contaminated sediment. EPA (1988) provides dredging cost estimates for use of hydraulic equipment as well as presents a generic cost range applicable to various sediment removal equipment types. More recent costs of dredging in the Great Lakes are provided by Averett et al. (1990). The costs presented here are within the cost ranges identified in the other sources.

4.2.10.2.3.3 Sediment Capping

Costs related to sediment containment with use of capping material are discussed in Section 4.2.7.2.3.3. There are no reported costs of this restoration option for lacustrine habitats; however, the costs reported for use of this action in estuarine and marine environments are generally applicable to lake systems.

4.3 Biological Natural Resource Restoration

Costs of restoration actions relating directly to fish and wildlife are discussed in the following sections. Each of the following five subsections: summarizes the oil discharge related and non-oil discharge related fish and wildlife restoration cost literature; provides the cost estimates of restoration actions; and discusses any assumptions or considerations related to the cost estimates of each option for shellfish (Section 4.3.1), fish (Section 4.3.2), reptiles (Section 4.3.3), birds (Section 4.3.4), and mammals (Section 4.3.5).

Exhibit 4.12 Reported costs for subtidal lacustrine restoration ($\$/m^3$ in mid-1992 dollars).

Sediment Removal		
Source	**Reported Costs**	**Other Reported Data**
Peterson (1979)[3] Hydraulic Dredging Bulldozer Dredging	 $0.53 - $6.20 $1.75 - $30.78	Average by Region: Great Lakes $ 2.63 Northwest $ 4.63 Central States $ 4.88 Northeast $11.04
Peterson (1982)[1]	$1.02 - $5.24	Central States Region
Cooke (1983)[1]	$0.30 - $19.00	Contaminated Soil >$34.00
EPA (1988)[1] Hydraulic Dredging General Dredging (Type not specified)	 $2.20 - $3.08 $0.42 - $24.79	 Cost range for 64 projects
Averett, et.al (1990)[4] Hydraulic Dredging Mechanical Dredging Hopper Dredging	 $5.66 - $11.34 $9.73 - $10.60 $5.01 - $7.57	 Contaminated Sediment Removal in Great Lakes Region

3 Reported costs do not include transport, disposal or monitoring costs.

4 Reported costs include transport and monitoring costs.

Natural recovery, or no action, allows injured environments to recover through natural processes. This action is typically used when no restoration alternatives exist or the alternatives would cause more injury than improvement. Natural recovery does require periodic monitoring of the area to ensure that adequate progress and recovery are occurring as expected (see Chapter 3). Restocking expedites the recovery process by introducing, or stocking, species the same as or comparable to those injured.

4.3.1 Shellfish

Shellfish resource restoration literature only provides cost estimates for mollusc reef restoration. A complete discussion of the economic costs related to this restoration is located in Section 4.2.4 (Mollusc Reefs). The following restoration actions are considered in Section 4.2.4:

- Natural Recovery;
- Reef Reconstruction; and
- Seeding of Beds.

Hatchery and seeding programs exist for other types of shellfish; however, these are currently only conducted in laboratory situations and were not established on a commercial basis (Exxon Valdez Oil Spill Trustees, 1992a; Lewis, 1993).

4.3.2 Fish

There exist five technically feasible actions for restoring injured fish habitats and populations. These actions include:

- Natural Recovery;
- Restocking/Replacement;
- Fishery Habitat Restoration and Enhancement;
- Modification of Fishery Management Practices; and
- Habitat Protection and Acquisition.

4.3.2.1 Oil Related Literature

After an extensive search of oil related restoration literature, no sources were located that discussed the costs of restoring fish populations to baseline levels.

4.3.2.2 Non-oil Related Literature

The most recent and comprehensive source of economic values of fish populations is a handbook published by the American Fisheries Society (Riely, Southwick, and Reilly, 1990). Part I of the handbook provides several valuation techniques for evaluating the economic damages resulting from a fish-kill event. The handbook includes replacement costs on a national and regional level, when available, for more than 100 marine and freshwater species, calculated based on a survey of the nation's public and private fish hatcheries.

In 1978, Nelson, Horak, and Olson of Enviro Control, Inc. prepared a handbook sponsored by the U.S. Department of the Interior that summarizes almost 300 fish and wildlife habitat and population improvement techniques. The actions discussed include enhancement techniques proven effective during previous dam and reservoir projects or determined potentially effective by experts in the field. A brief summary of each action provides relative costs among other information, such as engineering features, hydrological effects, biological effects, and related references. The fish and wildlife habitat improvement techniques are divided into several categories: reservoir flood basins; reservoir conservation pools; dam discharge systems; streamflows, riffles, and pools; streamside protection; and general practices. The fish and wildlife population improvement techniques are divided into the following categories: fish propagation; fish passage; fish stocking and control; wildlife propagation and control; and wildlife protection at canals.

Bell et al. (1989) evaluate the biological, physical, and economic effectiveness of eight manufactured artificial reef structures. These structures were tested at sites off the coast of South Carolina as part of the state's Marine Artificial Reef Program. Although the evaluation is ongoing to assess long-term effects, observation within the first three years of the study led to several preliminary conclusions and recommendations. Bell et al. describe the background of South Carolina's Marine Artificial Reef Program, methodology used for this study, specifications of the eight manufactured reef structures tested, economic cost of each reef structure type, and the preliminary results and conclusions of the study.

Prince and Maughan (1978) present and discuss several biological and economic issues relevant to the development of freshwater artificial reefs. The biological issues addressed include fish abundance, fish colonization, fish harvest rates, and fish production in freshwater environments in relation to the existence of artificial reefs. The discussion on economic issues emphasized the possibility of using donated equipment, supplies, and labor to construct artificial reefs. This discussion was based on an actual artificial reef development program for Smith Mountain Lake in Virginia.

Feigenbaum et al. (1989) discuss methodologies, results, and conclusions from a three-year artificial reef study program in the Chesapeake Bay supported by a mitigation fund. The study experimented with various reef structures and sites. The stress levels and stability of the structures were tested by placing them in both the bay and nearby coastal waters. Feigenbaum et al. (1989) also present success rates of the various reef structures and sites for attracting fish populations and increasing catch rates. Recommendations of the best structural types and reef locations were made based on the results of the study.

Knatz (1987) describes three projects under consideration as mitigation for port landfill development in Southern California. One project consists of constructing an artificial reef near the Port of Long Beach under the guidelines of state and federal wildlife agencies. The other two projects under consideration are wetland habitat enhancement projects near the port. The determination of adequate mitigation of a development project and the concept of mitigation banking are discussed. The relative technical concerns and cost estimates are provided for each project.

McGurrin and Fedler (1989) evaluate the planning, siting, and socio-economic impacts associated with the rigs-to-reefs development program, specifically the Tenneco II artificial reef project. This project consisted of transporting three obsolete petroleum platforms from Louisiana to south Florida. The platforms now serve as a large artificial reef site for recreational fishermen.

Smallowitz (1989) discusses the effects that the increasing number of hydroelectric dams in the Northwest have had on the annual runs of salmon and trout. The program to alleviate the injury inflicted on these migrating fish populations was initiated by the Northwest Power Act. The program includes both the enforcement of management practice policies and installation of mechanical fish passageways around or through the dams.

Watt (1986) describes a small liming program established to reduce the effects of acidity on the salmon populations which inhabit several rivers in Nova Scotia. This action could be applicable as off-site mitigation for a discharge. Chemical transportation on the rivers has caused the pH to decline. The restoration action presented as technically feasible in this situation is the addition of limestone to the rivers to counteract the acidic contamination. This same action can also be used on streams and lakes with low pH levels. In addition to describing the liming process, the estimated costs related to this effort and the expected benefits from the liming are presented and discussed.

4.3.2.3 Cost of Restoration Actions

The following paragraphs discuss the estimated costs involved with restoration of fish populations injured or destroyed by hazardous substance contamination. The actions include:

- Natural Recovery;
- Restocking/Replacement;
- Fishery habitat restoration and enhancement;
- Modification of fishery management practices; and
- Habitat protection and acquisition.

4.3.2.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.3.2.3.2 Restocking/Relocation

Two processes must be completed before restocking or relocation can occur. First, the habitat must be restored and free from contamination enough to support any fish or wildlife species reintroduced into the environment. Second, an assessment of the lost fish and wildlife species must be conducted in order to determine the related costs for this restoration option.

This section provides estimated costs associated with the process of restocking various fish species. The costs were obtained from the literature summarized above and are presented here on a per unit basis. These costs and the associated assumptions or considerations are discussed below.

There exist two principal methods of restocking. The trustees can either obtain fish from an established fish hatchery, assuming comparable fish species are readily available from a hatchery and within transportable distance from the restoration area, or develop hatcheries specifically for the purpose of restocking.

The American Fisheries Society handbook (Riely, Southwick, and Reilly, 1990) provides the costs of fish if obtained directly from a hatchery. These costs were developed from a survey of private and public hatcheries throughout the United States. The costs are provided either per fish, per pound of fish, or per inch of fish and are presented in Exhibits 4.13 through 4.19. The costs were calculated, when possible, for each U.S. Fish and Wildlife Service region. Exhibit 4.13 presents the fish values on a national level. When determining costs for specific species, however, the regional tables (Exhibits 4.14 through 4.19) should be consulted first. If the particular species is not listed in the table or the price is not available on a regional level, then the national table should be checked. The six U.S. Fish and Wildlife Service regions are listed below along with the table number that lists the costs for that region and states included in the region.

FWS Region	Exhibit #	States Included
1	4.14	HI, ID, NV, CA, OR, WA
2	4.15	AZ, NM, OK, TX
3	4.16	IL, IN, IA, MI, MN, MO, OH, WI
4	4.17	AL, AR, FL, GA, KY, LA, MS, NC, SC, TN
5	4.18	CT, DE, ME, MD, MA, NH, NJ, NY, PA, RI, VT, VA, WV
6	4.19	CO, KS, MT, NE, ND, SD, UT, WY

Alaska is Region 7, but there are not enough hatcheries from which to determine regional replacement costs.

This handbook also provides general transportation costs for transporting the fish from the hatchery to the point of release. Based on the survey of hatcheries, the average transportation cost is $1.20 per mile.

The U.S. Fish and Wildlife habitat and population improvement handbook (Nelson et al., 1978) provides costs associated with the development of a fish hatchery. The average capital cost of hatchery-raised fish, based on data from three projects and amortized over 20 years, is $2.35 (in mid-1992 dollars) per pound. The annual operation and maintenance cost of fish released from the hatchery, based on data from four projects, is $2.16 (in mid-1992 dollars) per pound. The total annual cost to operate a fish hatchery is thus $4.51 per pound of fish released.

Exhibit 4.13 Estimated costs of restocking various fish species in all U.S. Fish and Wildlife Service Regions (in mid-1992 dollars)

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
ACIPENSERIFORMES																
Acipenseridae (Sturgeons)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Acipenser oxyrhynchus (Atlantic sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Acipenser medirostris (Green sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Acipenser fulvescens (Lake sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Scaphirhynchus albus (Pallid sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Acipenser brevirostrum (Shortnose sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Scaphirhynchus platorynchus (Shovelnose sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Acipenser transmontanus (white sturgeon)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$80.28/lb
Polyodontidae (Paddlefish)																
Polyodon spathula (Paddlefish)	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$1.50	$35.21/lb
LEPISOSTEIFORMES																
Lepisosteidae (Gars)																
Atractosteus spatula (Alligator gar)	$0.85	$0.85	$0.85	$0.85	$1.06	$1.06	$1.06	$1.49	$1.49	$2.13	$2.13	$2.13	$2.13	$3.19	$3.95	$4.70/lb
Lepisosteus platyrhincus (Florida gar)	$0.85	$0.85	$0.85	$0.85	$1.06	$1.06	$1.06	$1.49	$1.49	$2.13	$2.13	$2.13	$2.13	$3.19	$3.95	$4.70/lb
Lepisosteus osseus (Longnose gar)	$0.85	$0.85	$0.85	$0.85	$1.06	$1.06	$1.06	$1.49	$1.49	$2.13	$2.13	$2.13	$2.13	$3.19	$3.95	$4.70/lb
Lepisosteus platostomus (Shortnose gar)	$0.85	$0.85	$0.85	$0.85	$1.06	$1.06	$1.06	$1.49	$1.49	$2.13	$2.13	$2.13	$2.13	$3.19	$3.95	$4.70/lb
Lepisosteus oculatus (Spotted gar)	$0.85	$0.85	$0.85	$0.85	$1.06	$1.06	$1.06	$1.49	$1.49	$2.13	$2.13	$2.13	$2.13	$3.19	$3.95	$4.70/lb
AMIIFORMES																
Amlidae (Bowfin)																
Amia calva (Bowfin)	$0.31 per pound															
ANGUILLIFORMES																
Anguillidae (Freshwater eels)																
Anguilla rostrata (American eel)	$2.13															
OSTEOGLOSSIFORMES																
Hiodonidae (Mooneyes)																
Hiodon alosoides (Goldeye)	$0.12	$0.12	$0.12	$0.26	$0.26	$0.26	$0.49	$0.49	$0.73	$0.73	$0.82	$0.82	$0.95	$0.95	$0.95	$0.95
Hiodon tergisus (Moodeye)	$0.12	$0.12	$0.12	$0.26	$0.26	$0.26	$0.49	$0.49	$0.73	$0.73	$0.82	$0.82	$0.95	$0.95	$0.95	$0.95
SALMONIFORMES																
Salmonidae (Trouts)																
Salmo salar (Atlantic salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb.
Oncorhynchus tshawytscha (Chinook salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb
Oncorhynchus keta (Chum salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb
Oncorhynchus kisutch (Coho salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb
Oncorhynchus gorbuscha (Pink salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb
Oncorhynchus nerka (Sockeye salmon)	$0.28	$0.31	$0.38	$0.47	$0.50	$0.53	$0.55	$0.72	$0.87	$1.12	$1.46	$2.12	$2.66	$3.30	$3.99	$1.94/lb
Salvelinus alpinus (Arctic char)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Thymallus articus (Arctic grayling)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Coregonus spp. (Cisco)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Salvelinus fontinalis (Brook trout)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb

Order, Family and Species	Cost per fish (unless otherwise stated) by length of restocked fish.															
	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Salmo trutta (Brown trout)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Oncorhynchus clarki (Cutthroat trout)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Salvelinus namaycush (Lake trout)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Prosopium spp. (whitefish)	$0.16	$0.22	$0.31	$0.41	$0.44	$0.48	$0.71	$0.90	$1.14	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Oncorhynchus mykiss (Rainbow trout)	$0.12	$0.18	$0.20	$0.27	$0.34	$0.43	$0.57	$0.71	$0.95	$1.49	$1.70	$2.02	$2.33	$2.83	$2.93	$1.67/lb
Umbridae (Mudminnows)										$1.28	$1.38	$1.76	$2.20	$2.57	$2.89	$1.65/lb
Umbra spp. (Mudminnow)	$0.09 per pound															
Esocidae (Pikes)																
Esox niger Chain pickerel	$0.14	$0.30	$0.51	$0.85	$0.85	$0.85	$1.19	$1.19	$1.70	$1.70	$1.70	$1.70	$2.79 per pound			
Esox americanus vermiculatus (Grass pickerel)	$0.14	$0.30	$0.51	$0.85	$0.85	$0.85	$1.19	$1.19	$1.70	$1.70	$1.70	$1.70	$2.79 per pound			
Esox lucius (Northern pike)	$0.14	$0.30	$0.51	$0.85	$0.85	$0.85	$1.19	$1.19	$1.70	$1.70	$1.70	$1.70	$2.79 per pound			
Esox americanus americanus (Redfin pickerel)	$0.14	$0.30	$0.51	$0.85	$0.85	$0.85	$1.19	$1.19	$1.70	$1.70	$1.70	$1.70	$2.79 per pound			
Esox masquinongy (Muskellunge)	$1.28	$3.72	$6.38	$7.45	$9.31	$9.79	$12.62	$15.14	$16.76	$19.15	$33.14/per pound					
Esox lucius/masquinongy (Tiger muskellunge)	$1.28	$3.72	$6.38	$7.45	$9.31	$9.79	$12.62	$15.14	$16.76	$19.15	$33.14/per pound					
CYPRINIFORMES																
Characidae (Characins)																
Astyanax mexicanus (Mexican tetra)	$0.09															
Cyprinidae (Minnows and Carps)																
Cyprinus carpio (Common carp)	$0.09	$0.09	$0.09	$0.09	$0.09	$0.09	$0.16	$0.16	$0.19	$0.24	$0.29	$0.29 per pound				
Campostoma spp. (Stoneroller)	$0.09	$0.09	$0.09	$0.09	$0.09	$0.09	$0.16	$0.19	$0.24	$0.29	$0.29	$0.29 per pound				
Pimephales promelas (Fathead minnow)	$0.01															
Notemigonus crysoleucas (Golden shiner)	$0.20	$0.20	$0.20	$0.20	$0.30	$0.30	$3.46 per pound									
Cterlopharyrlgodon idella (Grass carp)	$0.37	$0.74	$0.74	$1.50	$2.62	$2.62	$2.62	$3.60					$3.60 per pound			
Other cyprinids	$0.09															
Ictiobus cyprinellus (Bigmouth buffalo)	$0.14	$0.14	$0.14	$0.14	$0.24	$0.24	$0.32	$0.32	$0.32	$0.44	$0.51	$0.63	$0.63 per pound			
Ictiobus niger (Black buffalo)	$0.14	$0.14	$0.14	$0.14	$0.24	$0.24	$0.32	$0.32	$0.32	$0.44	$0.51	$0.63	$0.63 per pound			
Ictiobus babalus (Smallmouth buffalo)	$0.14	$0.14	$0.14	$0.14	$0.24	$0.24	$0.32	$0.32	$0.32	$0.44	$0.51	$0.63	$0.63 per pound			
Hypentelium etowanum (Alabama hog sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma duquesnei (Black redhorse)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma poecilurum (Blacktail redhorse)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Cycleptus elongatus (Blue sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Erimyzon oblongus (Creek chubsucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma erythrurum (Golden redhorse)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Erimyzon sucetta (Lake chubsucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Catostomus catostomus (Longnose sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Catostomus platyrhynchus (Mountain sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Hypentelium nigricans (Northern hog sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma carinatum (River redhorse)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma macrolepidotum (shorthead redhorse,	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Moxostoma anisurum (Silver redhorse)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Minytrema melanops (spotted sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Catostomus comnersoni (w'hite sucker)	$0.37	$0.37	$0.85	$0.85	$0.85	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	$2.42 per pound		
Carpiodes cyprinus (Quillback)	$0.06	$0.06	$0.06	$0.07	$0.07	$0.07	$0.14	$0.14	$0.18	$0.18	$0.24	$0.29	$0.29 per pound			
Carpiodes carpio (River carpsucker)	$0.06	$0.06	$0.06	$0.07	$0.07	$0.07	$0.14	$0.14	$0.18	$0.18	$0.24	$0.29	$0.29 per pound			

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
SILURIFORMES																
Ictaluridae (Freshwater catfish)																
Ictalurus furcatus (Blue catfish)	$0.19	$0.23	$0.24	$0.26	$0.29	$0.41	$0.44	$0.52	$0.71	$0.77	$0.84	$1.20	$1.56	$1.96	$1.17 per pound	
Ictalurus punctatus (Channel catfish)	$0.19	$0.23	$0.24	$0.26	$0.29	$0.41	$0.44	$0.52	$0.71	$0.77	$0.84	$1.20	$1.56	$1.96	$1.17 per pound	
Pylodictus olivaris (Flathead catfish)	$0.19	$0.23	$0.24	$0.26	$0.29	$0.41	$0.44	$0.52	$0.71	$0.77	$0.84	$1.20	$1.56	$1.96	$1.17 per pound	
Ictalurus catus (white catfish)	$0.19	$0.23	$0.24	$0.26	$0.29	$0.41	$0.44	$0.52	$0.71	$0.77	$0.84	$1.20	$1.56	$1.96	$1.17 per pound	
Ictalurus melas (Black bullhead)	$0.53	$0.53	$0.80	$0.80	$1.33	$17.44 per pound										
Ictalurus nebulosus (Brown bullhead)	$0.53	$0.53	$0.80	$0.80	$1.33	$17.44 per pound										
Ictalurus platycephalus (Flat bullhead)	$0.53	$0.53	$0.80	$0.80	$1.33	$17.44 per pound										
Noturus spp. (Madtoms)	$0.53	$0.53	$0.80	$0.80	$1.33	$17.44 per pound										
Ictalurus natalis (Yellow bullhead)	$0.53	$0.53	$0.80	$0.80	$1.33	$17.44 per pound										
Aphredoderus sayanus (Pirate perch)	$0.09															
Percopsis omiscomaycus (Trout-perch)	$0.09															
ATHERINIFORMES																
Cyprinidonitae (Killifishes)																
Fundulus spp.(Killifish, topminnows, studfish	$0.09															
Poeciliidae (Livebearers)																
Gambusia affinis (Mosquitofish)	$0.09															
Atherinidae (Silversides)																
Labidesthes sicculus (Brook silverside	$0.09															
Menidia beryllina (Inland silverside)	$0.09															
Menidia extensa (Waccamaw silverside)	$0.09															
GASTEROSTEIFORMES																
Gasterosteidae (Sticklebacks)																
Apeltes quadracus (4-spine stickleback)	$0.09															
Gasterosteus aculeatus (3-spine stickleback)	$0.09															
PERCIFORMES																
Percichthyidae (Temperate basses)																
Morone saxatilis (Striped bass)	$0.17	$0.31	$0.47	$0.64	$0.92	$0.92	$1.24	$1.46	$1.65	$1.82	$2.13	$2.73	$2.60 per pound			
Morone chrysops (white bass)	$0.16	$0.32	$0.48	$0.64	$0.80	$0.96	$1.12	$1.44	$1.44	$1.60	$1.76	$1.92	$1.22 per pound			
Morone mississippiensis (Yellow bass)	$0.16	$0.32	$0.48	$0.64	$0.80	$0.96	$1.12	$1.44	$1.44	$1.60	$1.76	$1.92	$1.22 per pound			
Monone americana (white perch)	$0.12	$0.19	$0.32	$0.40	$0.60	$0.70	$0.82	$0.95	$1.06	$1.18	$1.31	$1.40	$1.40 per pound			
Centrarchidae (Sunfishes)																
Micropterus salmoides (Largemouth bass)	$0.24	$0.34	$0.55	$0.76	$1.34	$1.67	$2.42	$2.97	$3.71	$3.98	$4.23	$4.28	$4.12 per pound			
Micropterus coosae (Redeye bass)	$0.24	$0.34	$0.55	$0.76	$1.34	$1.67	$2.42	$2.97	$3.71	$3.98	$4.23	$4.28	$4.12 per pound			
Micropterus punctulatus (Spotted bass)	$0.24	$0.34	$0.55	$0.76	$1.34	$1.67	$2.42	$2.97	$3.71	$3.98	$4.23	$4.28	$4.12 per pound			
Micropterus dolomieui (Smallmouth bass)	$0.70	$0.70	$1.45	$1.64	$2.14	$2.63	$3.17	$5.09	$6.54	$6.54	$8.51	$8.51	$8.51	$4.77 per pound		
Pomoxis nigromaculatus (Black crappie)	$0.16	$0.29	$0.39	$0.49	$0.73	$0.76	$0.90	$1.22	$3.92 per pound							
Pomoxis annularis (white crappie)	$0.16	$0.29	$0.32	$0.44	$0.73	$0.76	$0.90	$1.22	$3.92 per pound							
Felassoma zonatum (Banded pygmy sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Enneacanthus obesus (Banded sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis macrochirus (Bluegill)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Enneacanthus gloriosus (Bluespotted sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis marginatus (Dollar sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						

Order, Family and Species	Cost per fish (unless otherwise stated) by length of restocked fish.															
	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Centrarchus macropterus (Flier)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis cyanellus (Green sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis megalotis (Longear sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis humilis (Orangespotted sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis gibbosus (Pumpkinseed)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis auritus (Redbreast sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis microlophus (Redear sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Ambloplites rupestris (Rock bass)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Ambloplites ariommus (Shadow bass)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis punctatus (Spotted sunfish)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Lepomis gulosus (warmouth)	$0.17	$0.20	$0.32	$0.44	$0.59	$0.88	$0.92	$1.19	$1.60	$2.43 per pound						
Perca flavenscens (Yellow perch)	$0.30	$0.49	$0.80	$1.03	$1.45	$1.59	$2.00	$9.57 per pound								
Etheostoma spp.; Percina spp. (Darters)	$0.30	$0.49	$0.80	$1.03	$1.45	$1.59	$2.00	$9.57 per pound								
Stizostedion canadense (Sauger)	$0.27	$0.36	$0.81	$0.99	$1.54	$1.94	$2.67	$3.02	$3.30	$3.69	$4.56	$5.55	$9.21	$11.40	$13.78	$6.69/lb
Stizostedion vitreum vitreum (walleye)	$0.27	$0.36	$0.81	$0.99	$1.54	$1.94	$2.67	$3.02	$3.30	$3.69	$4.56	$5.55	$9.21	$11.40	$13.78	$6.69/lb
Aplodinotus grunniens (Freshwater drum)	$0.12			$0.19			$0.29		$0.40			$0.48	$0.55	$0.55 per pound		
Cichlidae (Cichlids)																
Tilapia melanotheron (Blackchin tilapia)	$0.07	$0.07	$0.15	$0.15	$0.22	$0.22	$0.27	$0.30	$0.33	$0.37	$0.41	$0.45	$0.45 per pound			
Tilapia aurea (Blue tilapia)	$0.07	$0.07	$0.15	$0.15	$0.22	$0.22	$0.27	$0.30	$0.33	$0.37	$0.41	$0.45	$0.45 per pound			
Tilapia mossambica (Mozambique tilapia)	$0.07	$0.07	$0.15	$0.15	$0.22	$0.22	$0.27	$0.30	$0.33	$0.37	$0.41	$0.45	$0.45 per pound			
Tilapia zilli (Redbelly tilapia)	$0.07	$0.07	$0.15	$0.15	$0.22	$0.22	$0.27	$0.30	$0.33	$0.37	$0.41	$0.45	$0.45 per pound			
Tilapia mariae (Spotted tilapia)	$0.07	$0.07	$0.15	$0.15	$0.22	$0.22	$0.27	$0.30	$0.33	$0.37	$0.41	$0.45	$0.45 per pound			
Cottidae (Sculpins)																
Cottus spp. (Sculpin)	$0.09															

* For Fish longer than 30 inches
Source: American Fisheries Society, 1990

Exhibit 4.14 Estimated costs of restocking various fish species in U.S. Fish and Wildlife Service Region 1 (in mid-1992 dollars).

	Cost per fish (unless otheriwse stated) by length of restock fish.															
Order, Family and Species	**1 in**	**2 in**	**3 in**	**4 in**	**5 in**	**6 in**	**7 in**	**8 in**	**9 in**	**10 in**	**11 in**	**12 in**	**13 in**	**14 in**	15 in	over 15 inches
SALMONIFORMES																
Salmonidae (Trouts)																
Salmo salar (Atlantic salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus tsha~ytscha (Chinook salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus keta (Chum salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus kisutch (Coho salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus gorbuscha (Pink salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus nerka (Sockeye salmon)	$0.28	$0.35	$0.38	$0.49	$0.52	$0.55	$0.57	$0.65	$0.95	$1.04	$1.37	$1.78	$2.23	$2.76	$3.34	N/A
Oncorhynchus mykiss (Rainbow trout)	$0.09	$0.12	$0.17	$0.22	$0.30	$0.33	$0.35	$0.53	$0.76	$1.29	$1.83	$2.36	$3.00	$3.81	$4.53	N/A

NA = The regional fish value is not available. Refer to Table 3.12 for the national value.
Source: American Fisheries Society, 1990

Exhibit 4.15 Estimated costs of restocking various fish species in U.S. Fish and Wildlife Service Region 2 (in mid-1992 dollars).

	Cost per fish (unless otherwise stated) by length of restocked fish.														
Order, Family, and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15	Over 15 in
SALMONIFORMES																
Salmonidae (Trouts)																
Salvelinus alpinus (Arctic char)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Thymallus articus (Arctic grayling)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Coregonus spp. (Cisco)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Salvelinus fontinalis (Brook trout)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Salmo trutta (Brown trout)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Oncorhynchus clarki (Cutthroat trout)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Salvelinus namaycush (Lake trout)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Prosopium spp. (whitefish)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
Oncorhynchus mykiss (Rainbow trout)	$0.13	$0.26	$0.36	$0.49	$0.62	$0.74	$0.87	NA	NA	NA	NA	NA	NA	NA	NA	NA
CYPRINIFORMES																
Cyprinidae (Minnows and Carps)																
Hypentelium etowanum (Alabama hog sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma duquesnei (Black redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma poecilurum (Blacktail redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Cycleptus elongatus (Blue sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Erimyzon oblongus (Creek chubsucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma erythrurum (Golden redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Erimyzon sucetta (Lake chubsucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Catostomus catostomus (Longnose sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Catostomus platyrhynchus (Mountain sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Hypentelium nigricans (Northern hog sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma carinatum (River redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma macrolepidotum (Shorthead redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Moxostoma anisurum (Silver redhorse)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Minytrema melanops (Spotted sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA
Catostomus commersoni (white sucker)	NA	NA	$1.06	$1.06	$1.06	$1.06	$2.13	$2.13	$2.13	$2.13	$2.13	$2.13	$3.19	NA	NA	NA

Exhibit 4.16 Estimated costs of restocking various fish species in U.S. Fish and Wildlife Service Region 3 (in mid-1992 dollars).

	Cost per fish (unless otherwise stated) by length of restocked fish.															
	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
SALMONIFORMES																
Salmonidae (Trouts)																
Salmo salar (Atlantic salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Oncorhynchus tshawytscha (Chinook salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Oncorhynchus keta (Chum salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Oncorhynchus kisutch (Coho salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Oncorhynchus gorbuscha (Pink salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Oncorhynchus nerka (Sockeye salmon)	NA	$0.09	$0.28	$0.29	$0.31	$0.55	$0.63	$0.81	$1.81	$2.45	$3.21	$4.13	$5.27	$6.56	$7.92	NA
Salvelinus alpinus (Arctic char)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Thymallus articus (Arctic grayling)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Coregonus spp. (Cisco)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Salvelinus fontinalis (Brook trout)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Salmo trutta (Bro~n trout)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Oncorhynchus clarki (Cutthroat trout)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Salvelinus namaycush (Lake trout)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Prosopium spp. (Whitefish)	$0.21	$0.26	$0.30	$0.34	$0.38	$0.43	$0.50	$0.65	$0.81	$1.05	$1.20	$1.23	$1.56	$1.95	$2.31	NA
Oncorhynchus mykiss (Rainbow trout)	$0.11	$0.16	$0.17	$0.22	$0.29	$0.45	$0.46	$0.66	$0.88	$1.11	$1.33	$1.55	$1.79	$2.05	$3.17	NA
Esocidae (Pikes)																
Esox niger Chain pickerel	$0.16	$0.27	$0.51	$0.85	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox americanus vermiculatus (Grass pickerel)	$0.16	$0.27	$0.51	$0.85	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox lucius (Northern pike)	$0.16	$0.27	$0.51	$0.85	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox americanus americanus (Redfin pickerel)	$0.16	$0.27	$0.51	$0.85	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox masquinongy (Muskellunge)	$1.60	$3.19	$5.52	$6.38	$7.98	$8.38	$10.33	$12.40	$14.36	$17.02	NA	NA	NA	NA	NA	NA
Esox lucius/masquinongy (Tiger muskellunge)	$1.60	$3.19	$5.52	$6.38	$7.98	$8.38	$10.33	$12.40	$14.36	$17.02	NA	NA	NA	NA	NA	NA
SILURIFORMES																
Ictaluridae (Freshwater catfish)																
Ictalurus furcatus (Blue catfish)	$0.03	$0.14	$0.19	$0.19	$0.26	$0.46	$0.51	$0.56	$0.66	$0.71	$0.79	$1.13	$1.47	$1.84	NA	NA
Ictalurus punctatus (Channel catfish)	$0.03	$0.14	$0.19	$0.19	$0.26	$0.46	$0.51	$0.56	$0.66	$0.71	$0.79	$1.13	$1.47	$1.84	NA	NA
Pylodictus olivaris (Flathead catfish)	$0.03	$0.14	$0.19	$0.19	$0.26	$0.46	$0.51	$0.56	$0.66	$0.71	$0.79	$1.13	$1.47	$1.84	NA	NA
Ictalurus catus (White catfish)	$0.03	$0.14	$0.19	$0.19	$0.26	$0.46	$0.51	$0.56	$0.66	$0.71	$0.79	$1.13	$1.47	$1.84	NA	NA
PERCIFORMES																NA
Percichthyidae (Temperate basses)																
Morone saxatilis (Striped bass)	$0.22	$0.65	$0.99	$2.50	$2.98	$2.98	$3.82	$3.90	$5.59	$5.59	NA	NA	NA	NA	NA	NA
Centrarchidae (Sunfishes)																
Micropterus salmoides (Largemouth bass)	$0.14	$0.36	$0.56	$0.66	$0.95	$1.34	$1.56	$2.22	$2.71	$2.71	$2.71	$2.71	NA	NA	NA	NA
Micropterus coosae (Redeye bass)	$0.14	$0.36	$0.56	$0.66	$0.95	$1.34	$1.56	$2.22	$2.71	$2.71	$2.71	$2.71	NA	NA	NA	NA
Micropterus punctulatus (Spotted bass)	$0.14	$0.36	$0.56	$0.66	$0.95	$1.34	$1.56	$2.22	$2.71	$2.71	$2.71	$2.71	NA	NA	NA	NA

	Cost per fish (unless otherwise stated) by length of restocked fish.														
	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Micropterus dolomieui (Smallmouth bass)	NA	$0.68	$1.03	$1.38	$1.52	$3.03	$3.03	$6.70	$6.70	$6.70	NA	NA	NA	NA	NA	NA
Pomoxis nigromaculatus (Black crappie)	$0.11	$0.35	$0.39	$0.48	$0.53	$0.85	$0.90	$1.22	NA	NA	NA	NA	NA	NA	NA	NA
Pomoxis annularis (white crappie)	$0.11	$0.35	$0.39	$0.48	$0.53	$0.85	$0.90	$1.22	NA	NA	NA	NA	NA	NA	NA	NA
Felassoma zonatum (Banded pygmy sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus obesus (Banded sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis macrochirus (Bluegill)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus gloriosus (Bluespotted sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis marginatus (Dollar sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Centrarchus macropterus (Flier)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis cyanellus (Green sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis megalotis (Longear sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis humilis (Orangespotted sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gibbosus (Pumpkinseed)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis auritus (Redbreast sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis microlophus (Redear sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Ambloplites rupestris (Rock bass)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Ambloplites ariommus (Shadow bass)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis punctatus (Spotted sunfish)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gulosus (Warmouth)	$0.20	$0.27	$0.32	$0.38	$0.49	$0.72	$0.86	$1.05	NA	NA	NA	NA	NA	NA	NA	NA
Perca flavenscens (Yellow perch)	NA	$0.37	$0.46	$0.53	$0.53	$0.80	$0.80	NA	NA	NA	NA	NA	NA	NA	NA	NA
Etheostoma spp.; Percina spp. (Darters)	NA	$0.37	$0.46	$0.53	$0.53	$0.80	$0.80	NA	NA	NA	NA	NA	NA	NA	NA	NA
Stizostedion canadense (Sauger)	$0.13	$0.33	$0.79	$0.95	$1.27	$1.35	$1.44	$1.89	$2.27	$2.53	$3.66	$4.61	$9.21	$11.40	$13.78	NA
Stizostedion vitreum vitreum (Walleye)	$0.13	$0.33	$0.79	$0.95	$1.27	$1.35	$1.44	$1.89	$2.27	$2.53	$3.66	$4.61	$9.21	$11.40	$13.78	NA

* For Fish longer than 30 inches
Source: American Fisheries Society, 1990

Exhibit 4.17 Estimated costs of restocking various fish species in U.S. Fish and Wildilfe Service Region 4 (in mid-1992 dollars).

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
SALMONIFORMES																
Salmonidae (Trouts)																
Salvelinus alpinus (Arctic char)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Thymallus articus (Arctic grayling)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Coregonus spp. (Cisco)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Salvelinus fontinalis (Brook trout)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Salmo trutta (Brown trout)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Oncorhynchus clarki (Cutthroat trout)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Salvelinus namaycush (Lake trout)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Prosopium spp. (whitefish)	NA	$0.01	$0.02	$0.07	$0.10	$0.18	$0.27	$0.27	$0.33	$0.39	$0.52	$0.65	$0.81	$1.01	$1.22	NA
Oncorhynchus mykiss (Rainbow trout)	$0.09	$0.14	$0.15	$0.19	$0.24	$0.29	$0.35	$0.44	$0.54	$0.78	$0.90	$1.05	$1.47	$1.86	$2.25	NA
SILURIFORMES																
Ictaluridae (Freshwater catfish)																
Ictalurus furcatus (Blue catfish)	$0.02	$0.03	$0.07	$0.13	$0.15	$0.15	$0.16	$0.17	$0.22	$0.27	$0.38	$0.50	$0.56	$0.70	NA	NA
Ictalurus punctatus (Channel catfish)	$0.02	$0.03	$0.07	$0.13	$0.15	$0.15	$0.16	$0.17	$0.22	$0.27	$0.38	$0.50	$0.56	$0.70	NA	NA
Pylodictus olivaris (Flathead catfish)	$0.02	$0.03	$0.07	$0.13	$0.15	$0.15	$0.16	$0.17	$0.22	$0.27	$0.38	$0.50	$0.56	$0.70	NA	NA
Ictalurus catus (white catfish)	$0.02	$0.03	$0.07	$0.13	$0.15	$0.15	$0.16	$0.17	$0.22	$0.27	$0.38	$0.50	$0.56	$0.70	NA	NA
PERCIFORMES																
Percichthyidae (Temperate basses)																
Morone saxatilis (Striped bass)	$0.15	$0.33	$0.48	$0.64	$0.83	$0.92	$0.95	$1.02	$1.44	$1.60	$1.76	$1.92	NA			
Centrarchidae (Sunfishes)																
Micropterus salmoides (Largemouth bass)	$0.14	$0.23	$0.26	$0.35	$0.53	$0.66	$0.74	$0.92	$0.96	$1.06	$1.17	$1.28	NA			
Micropterus coosae (Redeye bass)	$0.14	$0.23	$0.26	$0.35	$0.53	$0.66	$0.74	$0.92	$0.96	$1.06	$1.17	$1.28	NA			
Micropterus punctulatus (Spotted bass)	$0.14	$0.23	$0.26	$0.35	$0.53	$0.66	$0.74	$0.92	$0.96	$1.06	$1.17	$1.28	NA			
Pomoxis nigromaculatus (Black crappie)	$0.07	$0.12	$0.16	$0.21	$0.27	$4.58	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Pomoxis annularis (white crappie)	$0.07	$0.12	$0.16	$0.21	$0.27	$4.58	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Felassoma zonatum (Banded pygmy sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus obesus (Banded sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis macrochirus (Bluegill)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus gloriosus (Bluespotted sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis marginatus (Dollar sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Centrarchus macropterus (Flier)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis cyanellus (Green sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis megalotis (Longear sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis humilis (Orangespotted sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gibbosus (Pumpkinseed)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis auritus (Redbreast sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA						
Lepomis microlophus (Redear sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Ambloplites rupestris (Rock bass)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Ambloplites ariommus (Shadow bass)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis punctatus (Spotted sunfish)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gulosus (warmouth)	$0.09	$0.11	$0.24	$0.40	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Stizostedion canadense (Sauger)	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Stizostedion vitreum vitreum (walleye)	$0.54	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

* For Fish longer than 30 inches
Source: American Fisheries Society, 1990

Exhibit 4.18 Estimated costs of restocking various fish species in U.S. Fish and Wildlife Service Region 5 (in mid-1992 dollars).

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
SALMONIFORMES																
Salmonidae (Trouts)																
Salvelinus alpinus (Arctic char)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Thymallus articus (Arctic grayling)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Coregonus spp. (Cisco)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Salvelinus fontinalis (Brook trout)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Salmo trutta (Brown trout)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Oncorhynchus clarki (Cutthroat trout)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Salvelinus namaycush (Lake trout)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Prosopium spp. (whitefish)	$0.13	$0.28	$0.37	$0.59	$0.66	$0.76	$1.05	$1.22	$1.51	$1.88	$2.23	$2.64	$2.88	$3.68	$4.28	NA
Oncorhynchus mykiss (Rainbow trout)	$0.12	$0.18	$0.20	$0.54	$0.62	$0.71	$0.97	$1.13	$1.39	$1.74	$1.83	$2.51	$2.92	$3.31	$3.77	NA
Esocidae (Pikes)																
Esox masquinongy (Muskellunge)	$2.13	$4.26	$6.38	$8.51	$10.64	$12.77	$14.90	$17.02	$19.15	$21.28	NA					
Esox lucius/masquinongy (Tiger muskellunge)	$2.13	$4.26	$6.38	$8.51	$10.64	$12.77	$14.90	$17.02	$19.15	$21.28	NA					
CYPRINIFORMES																
Cyprinidae (Minnows and Carps)																
Hypentelium etowanum (Alabama hog sucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma duquesnei (Black redhorse)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma poecilurum (Blacktail redhorse)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Cycleptus elongatus (Blue sucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Erimyzon oblongus (Creek chubsucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma erythrurum (Golden redhorse)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Erimyzon sucetta (Lake chubsucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Catostomus catostomus (Longnose sucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Catostomus platyrhynchus (Mountain sucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Hypentelium nigricans (Northern hog sucker)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma carinatum (River redhorse)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma macrolepidotum (shorthead redhorse,	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Moxostoma anisurum (Silver redhorse)	$0.37	$0.37	$0.64	$0.64	$0.64	NA	NA	NA	NA	NA	NA	NA	NA	NA		
Lepomis auritus (Redbreast sunfish)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Lepomis microlophus (Redear sunfish)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Ambloplites rupestris (Rock bass)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Ambloplites ariommus (Shadow bass)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Lepomis punctatus (Spotted sunfish)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Lepomis gulosus (warmouth)	$0.35	$0.35	$0.49	$0.65	$0.85	$1.03	$1.06	$1.60	$1.60	NA						
Perca flavenscens (Yellow perch)	NA	$0.59	$0.59	$0.69	$1.52	$2.36	$3.19	NA								
Etheostoma spp.; Percina spp. (Darters)	NA	$0.59	$0.59	$0.69	$1.52	$2.36	$3.19	NA								

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Stizostedion canadense (Sauger)	$0.69	$0.69	$0.90	$1.33	$3.19	$3.19	$6.38	$6.38	$6.38	$6.38	$6.38	$7.45	NA	NA	NA	NA
Stizostedion vitreum vitreum (walleye)	$0.69	$0.69	$0.90	$1.33	$3.19	$3.19	$6.38	$6.38	$6.38	$6.38	$6.38	$7.45	NA	NA	NA	NA

Exhibit 4.19 Estimated costs of restocking various fish species in U.S. Fish and Wildlife Service Region 6 (in mid-1992 dollars).

	Cost per fish (unless otherwise stated) by length of restocked fish.															
Order, Family and Species	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
SALMONIFORMES																
Salmonidae (Trouts)																
Salvelinus alpinus (Arctic char)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Thymallus articus (Arctic grayling)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Coregonus spp. (Cisco)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Salvelinus fontinalis (Brook trout)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Salmo trutta (Brown trout)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Oncorhynchus clarki (Cutthroat trout)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Salvelinus namaycush (Lake trout)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Prosopium spp. (whitefish)	NA	$0.15	$0.20	$0.27	$0.28	$0.30	$0.45	$0.63	$0.92	$0.98	$1.54	$1.95	$2.48	$3.11	NA	NA
Oncorhynchus mykiss (Rainbow trout)	$0.13	$0.20	$0.22	$0.29	$0.30	$0.38	$0.52	$0.66	$0.81	$1.02	$1.19	$1.60	$2.36	$2.96	$3.56	NA
Esocidae (Pikes)																
Esox niger Chain pickerel	$0.11	$0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox americanus vermiculatus (Grass pickerel)	$0.11	$0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox lucius (Northern pike)	$0.11	$0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Esox americanus americanus (Redfin pickerel)	$0.11	$0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
SILURIFORMES																
Ictaluridae (Freshwater catfish)																
Ictalurus furcatus (Blue catfish)	$0.07	$0.09	$0.10	$0.10	$0.11	$0.16	$0.17	$0.26	$0.36	$0.50	$0.66	$0.86	$1.27	$1.36	NA	
Ictalurus punctatus (Channel catfish)	$0.07	$0.09	$0.10	$0.10	$0.11	$0.16	$0.17	$0.26	$0.36	$0.50	$0.66	$0.86	$1.27	$1.36	NA	
Pylodictus olivaris (Flathead catfish)	$0.07	$0.09	$0.10	$0.10	$0.11	$0.16	$0.17	$0.26	$0.36	$0.50	$0.66	$0.86	$1.27	$1.36	NA	
Ictalurus catus (white catfish)	$0.07	$0.09	$0.10	$0.10	$0.11	$0.16	$0.17	$0.26	$0.36	$0.50	$0.66	$0.86	$1.27	$1.36	NA	
PERCIFORMES																
Centrarchidae (Sunfishes)																
Micropterus salmoides (Largemouth bass)	$0.11	$0.15	$0.30	$0.40	$0.71	$0.99	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Micropterus coosae (Redeye bass)	$0.11	$0.15	$0.30	$0.40	$0.71	$0.99	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Micropterus punctulatus (Spotted bass)	$0.11	$0.15	$0.30	$0.40	$0.71	$0.99	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Micropterus dolomieui (Smallmouth bass)	$0.11	$0.14	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Pomoxis nigromaculatus (Black crappie)	NA	$0.18	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Pomoxis annularis (white crappie)	NA	$0.18	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Felassoma zonatum (Banded pygmy sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus obesus (Banded sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis macrochirus (Bluegill)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Enneacanthus gloriosus (Bluespotted sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis marginatus (Dollar sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Centrarchus macropterus (Flier)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis cyanellus (Green sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis megalotis (Longear sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Order, Family and Species	Cost per fish (unless otherwise stated) by length of restocked fish.															
	1 in	2 in	3 in	4 in	5 in	6 in	7 in	8 in	9 in	10 in	11 in	12 in	13 in	14 in	15 in	over 15 in
Lepomis humilis (Orangespotted sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gibbosus (Pumpkinseed)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis auritus (Redbreast sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis microlophus (Redear sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Ambloplites rupestris (Rock bass)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Ambloplites ariommus (Shadow bass)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis punctatus (Spotted sunfish)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Lepomis gulosus (warmouth)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Stizostedion canadense (Sauger)	$0.11	$0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Stizostedion vitreum vitreum (walleye)	$0.07	$0.12	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

* For Fish longer than 30 inches
Source: American Fisheries Society, 1990

The fish replacement costs presented in the American Fisheries Society handbook (Riely et al., 1990) and in Exhibits 4.13 through 4.19 include only the cost of purchasing the fish from a hatchery. The transportation costs are calculated separately based on the per-mile cost stated above and should be added to the cost of obtaining the fish. The labor and other expenses related to the restocking activities, however, are not included in the handbook. These costs would need to be estimated by the hatchery conducting the operation.

The fish hatchery development and operation costs from the U.S. Fish and Wildlife Service handbook are stated per pound of fish released from the hatchery (Nelson et al., 1978). Although the capital costs were amortized for 20 years, this lump sum of $46.97 (in mid-1992 dollars) per pound of fish raised would still need to be presented at the time of implementation. An additional limitation on the use of these costs, besides the age of the data, is that the type of fish raised in these hatchery projects is unknown. Costs can vary substantially based on the type of fish grown.

4.3.2.3.3 Fishery Habitat Restoration and Enhancement

There are two fishery habitat enhancement actions for which the literature provides the costs associated with implementation. These methods include the construction of artificial reefs and improvement of fish passageways. A detailed discussion on each of these actions located in Section 2.3.2.3.3.

Several studies document the costs involved with constructing and deploying artificial reef structures. Exhibit 4.20 presents the structural dimensions, unit costs, and installation costs for 12 artificial reef structures. All of the cost estimates were converted into mid-1992 dollars. There exist several other artificial reef structure types for which the literature fails to provide estimated costs.

The most expensive artificial reef structure is the polyolefin plastic cone at approximately $7,400 per unit. This design is manufactured from 2 cm thick cross-linked polyolefin plastic resin specifically for the purpose of artificial reefs (Bell et al., 1989). Whereas, the triangle tire unit is the least expensive structure at $3.77 each because of the simple design, construction process, and low material requirements (Prince and Maughan, 1978). The installation costs presented in Exhibit 4.20 have some correlation between the relative cost and type of labor and equipment required for deployment. For example, the reef structures requiring specialized equipment (i.e., forklift, crane) and substantial manual labor will cost more to install than those units that require less complicated logistics (Bell et al., 1989).

Exhibit 4.20 Estimated costs of artificial reef structures (in mid-1992 dollars).

Type of Artificial Reef	Dimensions of Reef Structure (in (meters)	Cost per Reef Structure	Installation Cost per Reef Structure	Source
Brush Shelters	10.7 Diameters/6.1 H	$131.44	*	Nelson et al (1978)
Concrete Igloos	NA	$1,332.00	NA	Feigenbaum et al. (1989)
Concrete, Rubble, and Riprap Rock Pile	10.7 Diameter/3.05 H	$6,679.14	*	Knatz (1987)
Modified Concrete Docks and PVC Plastic	3.0L/1.5 W/1.8 H	$1,000.15	$150.63	Bell et al. (1989)
Polyolefin Plastic Cone	NA	$7,383.42	$248.60	Bell et al. (1989)
Polyolefin Plastic Hemisphere	1.8 Diameter	$1,777.49	$124.30	Bell et al. (1989)
1.1 m Steel Reinforced Concrete Pipe	0.9 Diameter/2.4 L	$223.52	$207.01	Bell et al. (1989)
1.6 m Steel Reinforced Concrete Pipe	1.4 Diameter/1.2 L	$236.18	$292.82	Bell et al. (1989)
Structural Stele Cube	1.5 L/1.5 W/1.5 H	$274.74	$53.02	Bell et al. (1989)
Modified Steel Cube and Plastic Mesh	1.5 L/1.5 W/1.5 H	$298.84	$48.20	Bell et al. (1989)
Tires-in-Concrete	1.9 L/1.5 W/1.1 H	$108.45	$106.04	Bell et al. (1989)
Tires-in-Concrete	0.61 L/0.76 W/0.15H	$8.88	NA	Feigenbaum et al. (1989)
Triangle Tire Unit	NA	$3.77	$1.65	Prince and Maughan (1978)

* The cost of installation is included in the reef structure cost.

Nelson et al. (1978) provide estimated costs associated with two fish passageway improvements techniques; trap and haul systems, and fishways. These costs are based on a compilation of actual case studies related to each passageway improvement option were inflated to represent mid-1992 dollars. The average cost of constructing and installing a trap and haul system, based on three actual projects, was $4.5 million, with operation and maintenance costs at approximately $52,000 per year. Fishways, which include various fish ladder designs, are substantially more expensive to develop. The average construction and installation cost, based on two actual project budgets, was $15.8 million.

4.3.2.3.4 Modification of Management Practices

The literature does not provide the costs associated with this restoration action.

4.3.2.3.5 Habitat Protection and Acquisition

The literature does not provide the costs associated with this restoration action.

4.3.3 Reptiles

There exist three actions for restoring injured reptile habitats and populations. These actions include:

- Natural Recovery;
- Restocking/Replacement; and
- Protection of Nest Sites.

4.3.3.1 Oil Related Literature

After an extensive search of oil related restoration literature, no sources were located that discussed the costs of restoring reptile populations to baseline conditions following an oil discharge incident.

4.3.3.2 Non-oil Related Literature

A report developed by International Animal Exchange, Inc. (1992) provides the cost estimates and availability to deliver live wildlife specimens from captive sources for the purpose of reintroduction to the wild in U.S. territories and estimated cost to obtain, transport, and acclimate wildlife specimens from the wild. The estimated costs of restocking an affected reptile population with captive raised reptiles or reptiles relocated from another location are discussed below in Section 4.3.3.3.2.

4.3.3.3 Cost of Restoration Actions

The following paragraphs discuss the estimated costs involved with restoration of reptiles injured or destroyed by oil contamination. The actions include:

- Natural Recovery;
- Restocking/replacement; and
- Protection of Nest Sites.

4.3.3.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.3.3.3.2 Restocking/Replacement

Before restocking or replacement can occur, the habitat must be restored and free from contamination enough to support any reptile species reintroduced into the environment.

This section provides estimated costs associated with the process of restocking or replacing various reptile species. These costs were obtained from the report provided by International Animal Exchange, Inc. (1992) and are presented on a per unit basis. These costs and the associated assumptions or considerations are discussed below.

According to the report by International Animal Exchange, Inc. (1992), there exist seven reptile species that can potentially be relocated from one location to another location in the wild. Of these seven, however, only one species, the American alligator, can be raised in captivity and released into the wild. The estimated costs per animal in mid-1992 dollars of both the captive raised alligator and species that can be relocated are provided in Exhibit 4.21. The costs for relocated reptiles range from $2,400 per specimen for an Atlantic loggerhead turtle to $7,800 per specimen for a Pacific ridley turtle. The cost of a captive raised American alligator is $2,600 per specimen.

There are several assumptions associated with the costs provided in the previous section: no licenses or permit fees are included in the stated cost; values for the animals relocated from another location in the wild are based on relocating a minimum of 10 specimens of a species; and costs include transportation, personnel, supplies, and equipment expenses.

Exhibit 4.21 Estimated costs of restocking reptiles (in mid-1992 dollars).

		Cost per animal	
Family	**Species**	**Captive Raised**	**Relocated from Wild**
Cheloniidae	Atlantic Loggerhead turtle	NA	$2,400
	Pacific Loggerhead turtle	NA	$2,800
	Atlantic ridley turtle	NA	$6,900
	Pacific ridley turtle	NA	$7,800
Dermochelyidae	Atlantic leatherback turtle	NA	$6,600
	Pacific Sea leatherback turtle	NA	$4,600
Crocodylidae alligatorinae	American Alligator	$2,600	$3,200

NA = Not Applicable
Note: Please refer to text for related assumptions and to Section 2.3.3 for the availability of captive raised reptiles.

Source: International Animal Exchange, 1992

4.3.3.3.3 Protection of Nest Sites

The literature does not provide the costs associated with this restoration option.

4.3.4 Birds

There exist five actions for restoring injured bird habitats and populations. These actions include:

- Natural Recovery;
- Restocking/Replacement;
- Habitat Restoration and Enhancement;
- Modification of Management Practices; and
- Habitat Protection and Acquisition.

4.3.4.1 Oil Related Literature

After an extensive search of oil related restoration literature, no sources were located which discussed the economic costs of restoring bird populations.

4.3.4.2 Non-oil Related Literature

As described in Section 4.3.3.2., International Animal Exchange, Inc. (1992) developed a report which provides cost estimates of relocating or restocking various wildlife species. A majority of the species included are birds. The costs associated are discussed in detail in Section 4.3.4.3.2.

4.3.4.3 Estimated Costs of Restoration Actions

The following paragraphs discuss the estimated costs involved with restoration of bird communities injured or destroyed by oil contamination. The actions include:

- Natural Recovery;
- Restocking/Replacement;

- Habitat Restoration and Enhancement;
- Modification of Management Practices; and
- Habitat Protection and Acquisition.

4.3.4.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.3.4.3.2 Restocking/Replacement

Before restocking or replacement of birds can occur, the habitat must be restored and free enough from contamination to support any bird species reintroduced into the environment.

This section provides estimated costs associated with the process of restocking or replacing various species of birds. These costs were obtained from the report provided by International Animal Exchange, Inc. (1992) and are presented on a per unit basis. These costs and associated assumptions or considerations are discussed below.

According to the report by International Animal Exchange, Inc. (1992), there exist 88 species of birds that can potentially be relocated from one location in the wild to another location. Of these 88 species, however, only 24 species can be raised in captivity and released into the wild. The estimated costs per bird in mid-1992 dollars of both the captive raised species and species that can be relocated are provided in Exhibit 4.22. The costs for relocated birds range from $200 per bird for gulls or terns to $3,800 per bird for American flamingos. The costs of obtaining captive raised birds range from $200 per bird for ducks to $4,400 per bird for American white pelicans.

There are several assumptions associated with the costs provided in the previous section: no licenses or permit fees are included in the stated cost; costs for the birds relocated from the wild to another location are based on relocating a minimum of 100 specimens of a species; and costs include transportation, personnel, supplies, and equipment expenses.

4.3.4.3.3 Habitat Restoration and Enhancement

The literature does not provide the costs associated with this restoration option. However, habitat restoration costs provided earlier in this Section may be applicable, depending on the particular actions performed.

Exhibit 4.22 Estimated costs of restocking birds (in mid 1992 dollars).

Family	Species	Cost per animal	
		Captive Raised	**Relocated from Wild**
Caviidae	Common loon	NA	$330
Podicipedidae	Horned grebe	NA	$370
	Red necked grebe	NA	$410
Domedeidae	Laysan albatross	NA	$810
	Black footed albatross	NA	$810
Procellariidae	Northern fulmar	NA	$790
	Japanese petrel	NA	$805
	Hawaiian petrel	NA	$690
	Greater shearwater	NA	$705
	Sooty shearwater	NA	$640
	Manx shearwater	NA	$640
	Shorttailed shearwater	NA	$640
Hydrobatidae	Least storm petrel	NA	$510
	White-vented storm petrel	NA	$510
	Band-rumped storm petrel	NA	$510
	Ashy storm petrel	NA	$510
	Ringed storm petrel	NA	$570
	Leachs storm petrel	NA	$690
Pelecanidae	American white pelican	$4,400	$2,400
	Brown pelican	$1,900	$920
Sulidae	Northern gannet	NA	$790
	Blue-footed booby	NA	$810
Phalacrocoracidae	Double crested cormorant	NA	$960
	SW Double crested cormorant	NA	$960
	NW Double-crested cormorant	NA	$960
	Common (great) cormorant	$710	$860
	Northern great cormorant	NA	$960
	Olivaceous cormorant	NA	$960
Ardeidae	American bittern	NA	$710
	Great blue heron	$2,200	$2,400
	Green heron	NA	$2,400
	Tricolored heron	NA	$2,400
	Black browned night heron	NA	$1,810
	Night heron	NA	$1,810
	Yellow-crowned night heron	NA	$1,810
	Cattle egret	$300	$305
	Snowy egret	$890	$405
Threskiornithidae	American white ibis	$690	$405
	Scarlet ibis	$700	$940
	Bare-faced ibis	NA	$640
	White-faxed ibis	NA	$640
	Glossy ibis	$590	$640
	Roseate spoonbill	$1,100	$1,400
Phoenicopteridae	American Flamingo	$2,100	$3,800
Anatidae	White-fronted goose	$600	$890
	Tule goose	NA	$850

	Graylag goose	$710	$850
	Snow goose	$450	$590
	Greater snow goose	NA	$710
	Lesser snow goose	$490	$710
	Emperor goose	$850	$975
	Ross goose	$710	$840
	Lawrences brant goose	NA	$840
	Pacific brant goose	NA	$840
	Canada goose (generic)	$360	$490
	Whistling swan	NA	$1,200
	Trumpeter swan	$1,200	$1,400
	Duck (most species; generic)	$200	$370
Accipitridae	Hawk/Eagle (most species; generic)	NA	$2,400
Gruidae	Whooping crane	NA	$1,640
	Sandhill Crane	$1,000	$1,340
	Lesser sandhill crane	$1,000	$1,340
	Florida sandhill crane	$1,000	$1,340
	Mississippi sandhill crane	NA	$1,640
	Canadian sandhill crane	NA	$1,640
	Greater sandhill crane	NA	$1,640
Aramidae	Limpkin	NA	$810
Rallidae	Rail/coot (most species; generic)	NA	$410
Haematopodidae	American oystercatcher	NA	$490
Recurvirostridae	Hawaiian stilt	NA	$490
	Black winged stilt	NA	$490
	Black necked stilt	NA	$490
	American avocet	NA	$590
Charadriidae	Lesser golden plover	NA	$590
	Black bellied plover	NA	$590
Scolopacidae	Spotted sandpiper	NA	$470
	Upland sandpiper	NA	$470
	Willet	NA	$470
	Wandering tattler	NA	$670
	Godwit	NA	$670
	Long-billed curlew	NA	$740
	Lesser yellowlegs	NA	$490
	Greater yellowlegs	NA	$490
	Solitary sandpiper	NA	$710
	Black turnstone	NA	$910
	Andean snipe	NA	$910
Laridae	Gull/Turn (most species; generic)	NA	$200
Alcidae	Puffin (most specific; generic)	$2,100	$1,960

NA = Not available
Note: Please refer to test for related assumptions and to Section 2.3.4 for the availability of captive raised birds.
Source: International Animal Exchange, 1992.

4.3.4.3.4 Modification of Management Practices

The literature does not provide the costs associated with this restoration option.

4.3.4.3.5 Habitat Protection and Acquisition

The literature does not provide the costs associated with this restoration option.

4.3.5 Mammals

There exist five actions for restoring injured mammal habitats and populations. These actions include:

- Natural Recovery;
- Restocking/Replacement;
- Habitat Restoration and Enhancement;
- Modification of Management Practices; and
- Habitat Protection and Acquisition.

4.3.5.1 Oil Related Literature

After an extensive search of oil related restoration literature, no sources were located which discussed the economic costs of restoring mammal populations.

4.3.5.2 Non-oil Related Literature

As described in Section 4.3.3.2., International Animal Exchange, Inc. (1992) developed a report which provides availability levels of various wildlife species. The estimated costs of restocking are discussed in detail in Section 4.3.5.3.2.

4.3.5.3 Estimated Cost of Restoration Actions

The following paragraphs discuss the estimated costs involved with restoration of mammals injured or destroyed by oil contamination through natural recovery and restocking/replacement.

4.3.5.3.1 Natural Recovery

Section 4.4 provides a description of the costs of monitoring programs.

4.3.5.3.2 Restocking/Replacement

Before restocking can be undertaken the habitat must be restored and free from contamination enough to support any mammal species reintroduced into the environment, and an assessment of the lost mammal species must be conducted to determine the related costs for this restoration option.

This section provides estimated costs associated with the process of restocking or relocating various species of mammals. These costs were obtained from the report provided by International Animal Exchange, Inc. (1992) and are presented on a per unit basis. These costs and associated assumptions or considerations are discussed below.

According to the report by International Animal Exchange, Inc. (1992), there exist 27 species of mammals that can potentially be relocated from one location in the wild to another location. Of these 27 species, however, only 7 species can be raised in captivity and released into the wild. The estimated costs per mammal in mid-1992 dollars of both the captive raised species and species that can be relocated are provided in Exhibit 4.23. The costs for relocated mammals range from $200 per specimen for muskrats to $235,000 per specimen for Northern right-whale dolphins. The costs of obtaining captive raised mammals range from $4,000 per specimen for Northern fur seals to $65,000 per specimen for bottle-nosed dolphins.

The assumptions associated with the costs provided in the previous section include: no licenses or permit fees are included in the stated cost; values for the animals relocated in the wild are based on relocating a minimum of 10 specimens of a species; and costs include transportation, personnel, supplies, and equipment expenses.

4.3.5.3.3 Habitat Restoration and Enhancement

The literature does not provide the costs associated with this restoration option. However, habitat restoration costs provided earlier in this section may be applicable, depending on the particular actions performed.

4.3.5.3.4 Modification of Management Practices

The literature does not provide the costs associated with this restoration action.

4.3.5.3.5 Habitat Protection and Acquisition

The literature does not provide the costs associated with this restoration action.

Exhibit 4.23 Estimated costs of restocking mammals (in mid-1992 dollars).

Family	Species	Captive Raise	Relocated from wild
Cricetidae	Muskrat	NA	$200
Delphinidae	Killer whale	NA	$160,000
	False killer whale	NA	$100,000
	Northern right-whale dolphin	NA	$235,000
	Saddle back dolphin	NA	$25,000
	Common dolphin	NA	$40,000
	Risso's dolphin	NA	$40,000
	White-sided dolphin	NA	$30,000
	Pacific white-sided dolphin	NA	$30,000
	Gill's bottle-nosed dolphin	NA	$40,000
	Bottle-nosed dolphin	$65,000	$35,000
	Pacific harbour porpoise	NA	$49,000
	Dall's porpoise	NA	$61,000
Monondontidae	Beluga whale	NA	$49,000
Ursidae	Polar bear	$20,000	$35,000
Mustelidae	Northern sea otter	$23,000	$13,000
	Southern sea otter	NA	$16,000
Otariidae	Northern fur seal	$4,000	$4,000
	Steller's northern sea lion	NA	$17,000
	California sea lion	$5,000	$4,000
	Walrus	NA	$39,000
	Bearded seal	NA	$24,000
	Grey seal	$5,000	$11,000
	Harbor seal	$5,000	$11,000
	Northern elephant seal	NA	$16,000
	Hawaiian monk seal	NA	$11,000
Trichechidae	Manatee	NA	$16,000

NA = not available
Note: Please refer to text for related assumptions and to Section 2.3.5 for the availability of captive raised mammals.
Source: International Animal Exchange, 1992

4.4 Monitoring Costs

In practice, a systematic monitoring program, with mechanisms to assess the effectiveness of the restoration strategy and, if necessary, make mid-course adjustments to that strategy is critical. The literature consistently recognizes the significance of implementing a reliable monitoring program as an integral part of the restoration process (Broome, 1990; Gore and Bryant, 1988; Josselyn et al., 1990; Lewis, 1990; and Nur and Ainley, 1992).

In establishing monitoring costs it must be remembered that monitoring program costs do not include those costs associated with environmental assessment activities associated with the damage assessment process. In a similar fashion, the analysis of the site which may be necessary to design and implement a restoration strategy are not monitoring costs. When the study and research costs associated with implementing a restoration option have been available, they have been captured in the unit costs of conducting the initial restoration activities presented in this section. The limited literature that directly estimates the costs associated with implementing a monitoring program are summarized below:

- **Artificial reef monitoring**. Knatz (1987) describes an artificial reef extension project in southern California. Due to the uncertainty of the project, the California Coastal Commission required the implementation of a two-year biological monitoring program. The costs of this program were estimated near $86,450 per hector (in 1992 dollars);

- **Liming program monitoring**. Watt (1986) describes a liming program in Nova Scotia. The objective of the program is to increase the pH level in several rivers to restore the declining Atlantic salmon stocks. The program duration is 20 years. The estimated cost for project management and scientific monitoring, which includes biological and chemical monitoring, is $621,500 per year (in 1992 dollars); and

- **Sediment monitoring.** The costs of sampling sediments for residual oil contamination are more readily available than those for other habitats because these tests are routinely preformed as part of the permitting process for dredging rivers and harbors. The collection of sediment samples for one area requires one to two days of sampling effort. Meyers et al. (1991) estimates sampling costs of $1,000 to $10,000 per day for boat rental plus an additional $1,000 to $3,000 per day for labor. For a generic sampling plan, Pequegnat et al. (1990) estimates vessel costs at approximately $8,700 per day, and scientist labor at $5,000 to $6,000. The New York/New Jersey Port Authority estimates their 1992 sampling costs at approximately $5,000 per mobilization and $1,500 per day of operation. Based on the above information, sampling costs for about five to eight sampling stations range from $2,000 to $15,000 per day. The number of samples required would vary substantially by the size of area being monitored, and the hydrological characteristics of the affected environment. Absent a large or complex sampling plan, one day of sampling effort would be adequate to cover a relatively large affected area of between 10 to 50 square miles

The more substantial costs are for analyzing and evaluating the results. The minimal test for organic pollutants average $1,200 per sample (EPA ERL-N 1991; Pequegnat et al, 1990) and simply represents the measurement of residual oil. Additional tests which could be required, depending on the nature of the concern, are as follows;

- The 96-hour elutriate bioassay with mysid shrimp is between $1,000 to $2,340 per sample (Meyers et al., 1991; NY/NJ Port Authority, 1992).

- 10-day benthic toxicity test with infaunal amphipod is between $400 and $4,207 per sample (Meyers et al., 1991; EPA ERL-N, 1991; NY/NJ Port Authority, 1992).

- 28-day bioaccumulation test, without chemical analysis of tissues, using a polychaete worm is $2,000 to $5,950 (Meyers et al., 1991; NY/NJ Port Authority, 1992).

Assuming five sediment samples and the simple bioassay analysis, monitoring costs could be as low as $5,000, but are expected to average approximately $20,000 per annum. If additional samples were taken and the more complex tests required, the total costs would be on the order of $40,000 to $125,000 per annum.

It should be cautioned that sediment monitoring tests would vary depending on the nature of the sediments, type of contamination and types of resources being monitored. Testing would at a minimum include testing for the simple presence of contamination. If the concern extended to the toxic effects on benthic species, more complex tests are available including a elutriate bioassay test with mysid shrimp, the benthic toxicity test using infaunal amphipod, or bioaccumulation tests with worms. These more complex tests would only be warranted in situations were there were substantial concerns over the contamination of sediments and represent an upper bound of costs.

5.1 Overview

In choosing among several restoration alternatives and actions for given habitats or natural biological resources (Exhibit 5.1), the following general approach should be considered:

- The baseline condition and functioning of the natural resource needs to be understood and quantified, including the degree of variability that exists. Degree of injury and natural recovery should be assessed;

- Incident- and natural resource-specific restoration goals and objectives are defined. The overall goal of restoration is to make the environmneta and public whole through the return of the injured natural resources and services to baseline and compensation of interim losses;

- Actions are evaluated for feasibility, i.e., whether actions are possible in the context of the particular situation. Constraints include availability of services, materials and equipment; construction and operational considerations; need or capability of future restoration; and consistency with all applicable laws and regulations. Infeasible actions are eliminated from further analysis. When practical, tested methods should take preference over unproven methods;

- The relative scientific merits (effectiveness) of feasible actions are evaluated;

- The most cost-effective actions that meet the restoration goals and objectives should be selected (i.e., if two or more actions provide equal benefits, the least costly is the most cost-effective action); and

- The expected costs of each action (or set of actions performed together) should be compared with expected benefits (where benefit estimation is feasible at a reasonable cost) to estimate reduction in interim loss.

In the following sections, an evaluation of actions for each habitat and natural resource is made that considers technical feasibility, scientific merit (effectiveness and success), and cost (Exhibit 5.2). Each of the possible restoration actions will be evaluated relative to the natural recovery alternative (no direct or primary action) and to all other feasible alternatives and actions for the habitat or resource. A system for selecting among alternatives and actions is developed that supports the decisionmaking framework (Chapter 6).

Exhibit 5.1 Restoration actions for each alternative.

1. Natural Recovery (no action)
   - Monitoring
2. Direct Restoration
   - a. Direct Habitat Restoration
     - Contaminant Removal
     - Reconstruction
     - Replanting
     - Accelerated Degradation
     - Monitoring
     - Maintenance
   - b. Direct Resource Restoration
     - Restocking
     - Harvest Alteration
     - Enhancement
     - Monitoring
     - Maintenance
3. Rehabilitation
   - a. Habitats
     - Contaminant Removal
     - Reconstruction
     - Replanting
     - Accelerated Degradation
     - Monitoring
     - Maintenance
   - b. Resource
     - Stocking
     - Harvest Alteration
     - Enhancement
     - Monitoring
     - Maintenance
4. Replacement
   - a. Habitats
     - Enhancement
     - Creation
     - Monitoring
     - Maintenance
   - b. Resources
     - Stocking
     - Harvest Alteration
     - Enhancement
     - Monitoring
     - Maintenance
   - c. Non-biological Services
     - Recreational
     - Commercial
     - Cultural
5. Acquisition of Equivalent Resources
   - Acquire property rights
   - Protection or management
6. Combinations of the Above

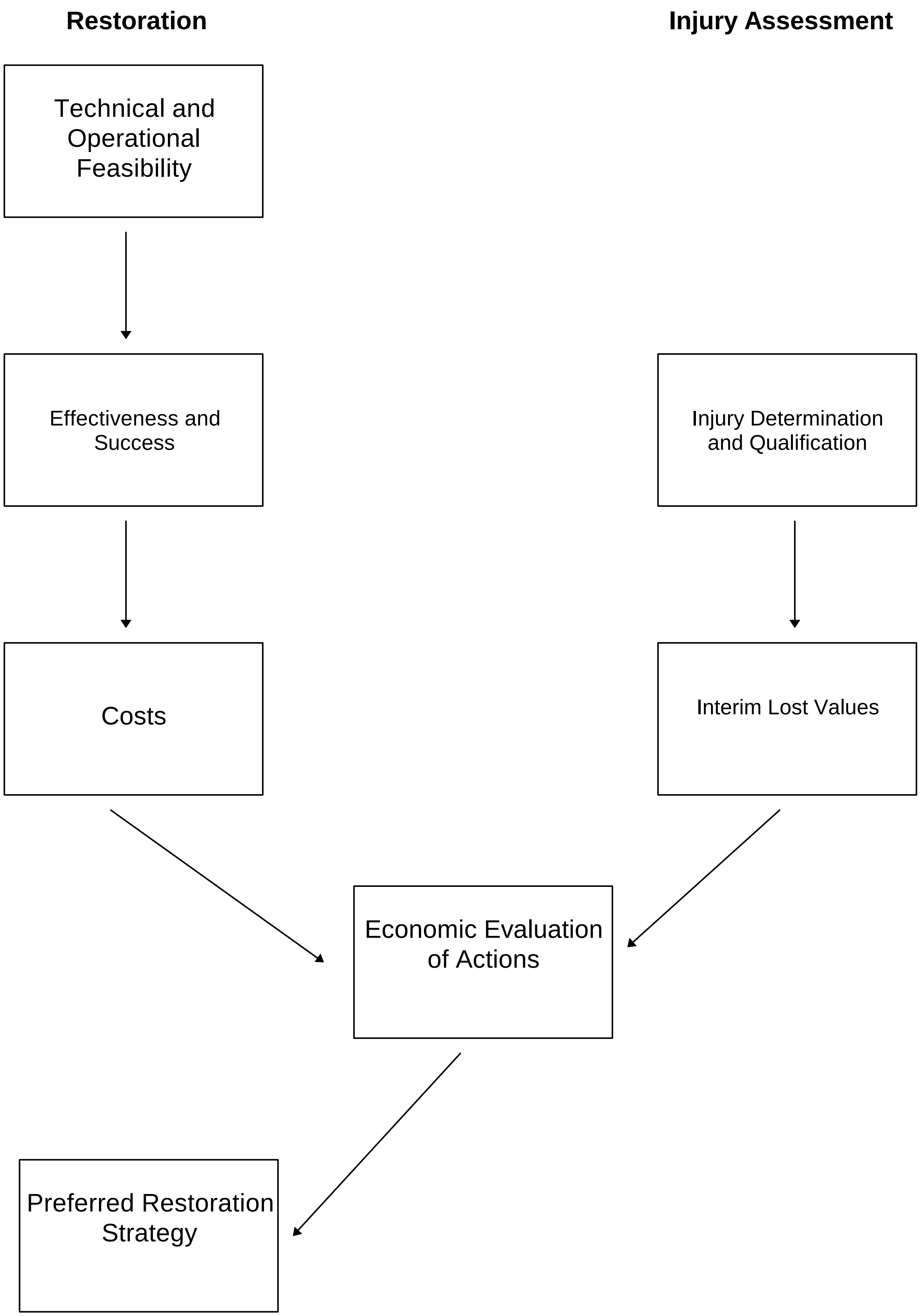

Exhibit 5.2 Process for recommending a restoration strategy.

In many cases, a qualitative assessment will show quite clearly that certain actions are either preferable or not viable. Also, the alternatives of direct restoration, rehabilitation, and replacement are preferable to acquisition of equivalent natural resources in the context of OPA. Thus, a first cut ranking system of actions may be made at this qualitative level. However, this evaluation should have some basis (i.e., feasibility or habitat recovery potential).

In addition to the assessment that must be assessed for a given natural resource, an evaluation among natural resources must also be made. While a particular action may not be effective at restoring the targeted resource, it may be of net benefit to all injured natural resources to perform that action. For example, cleaning oil off shorelines may be injurious to shoreline biota, but may reduce contamination effects on wildlife and other ecosystems.

Another aspect of the assessment involves the replacement of natural resources and their services by altering, and so impacting, other natural resources, for example in using wetland creation to replace affected wetlands and wildlife services. If an injured wetland is expected <u>never</u> to recover, then creation of two or more acres for every acre injured is appropriate. But if the injured wetland is expected to recover over some finite period, then a mitigation ratio of 2 or more might be over-compensating the public, if the created wetlands are expected to provide services in perpetuity. The total discounted flow of services in the created habitat should be just equal to the total discounted flow of services lost from the injured wetland. For more discussion of the methods for determining appropriate level of compensation, the reader is referred to the OPA regulations.

In his review of wetlands mitigation planning, Kruczynski (1989) makes the following points. The order of preference for mitigation (of wetlands loss) should be: (1) direct restoration of a degraded wetland (which may be other than the wetland injured), (2) creation of new wetland in an upland area not a wetland in the recent past, (3) enhancement of one or more functions of an existing wetland, (4) habitat exchange, which amounts to creating a wetland in an area which is presently a functional aquatic habitat of another type, and (5) preservation of existing habitat. He argues that choice (1) is more likely to be successful than choice (2). Both enhancement and exchange involve the replacement of some natural resources and services by others presumably more desirable. Preservation should not normally be considered compensatory for loss, since there is no net gain to the public. However, where preservation can be shown to prevent a future loss and where protection is in perpetuity this alternative may be a viable option.

Kruczynski (1989) also suggests compensatory mitigation ratios for wetlands to make up for the fact that restoration and replacement do not necessarily provide 100% of the services of natural wetlands (and in fact are really rehabilitation in the sense of the definitions used in this document). What is sought is functional equivalency to the wetland area injured. He suggests minimum ratios of 1.5:1 for restoration, 2:1 for creation, and 3:1 for enhancement, meaning that much more habitat should be restored, created, or enhanced to compensate for a unit loss of natural habitat. These ratios imply, however, that the converted habitat in the compensation (i.e., at the new site) is not of equivalent value to the (wetland) habitat created. These tradeoffs, need to be carefully evaluated.

5.1.1 Quantification of Recovery

In order to select the most appropriate restoration actions, quantitative information on the rate and level of recovery of natural resources and their services should be evaluated for each action and compared to other actions. As an illustration of this type of evaluation, a simple recovery model has been developed. An outline of the recovery model is as follows.

In the case of natural recovery, recovery is related to the concentration (or mass per unit area) of oil remaining in the habitat over time if that concentration is toxic. Analyses by Reed et al. (1989) have shown that for marine intertidal habitats (and others as well) concentration as a function of time may be described by a first-order decay curve, which may be written as:

$$\frac{dC}{dt} = -(d + r)C = -kC \qquad \textbf{(1)}$$

or

$$C = C_o \; e^{(-d-r)t} = C_o \; e^{-kt} \qquad \textbf{(2)}$$

where

- C = concentration (or mass per area)
- t = time
- C_o = C at t=0
- d = degradation rate
- r = physical (natural) removal rate
- k = d + r = decay constant measuring total loss rate

For some restoration actions, the values of d (e.g., bioremediation) or r (e.g., chemical remediation) are increased. Thus, C = f(t) may be described by changing the value of k at a certain time of restoration, t_r. For other actions (e.g., mechanical removal), a fraction of C is removed at t_r (Exhibit 5.3).

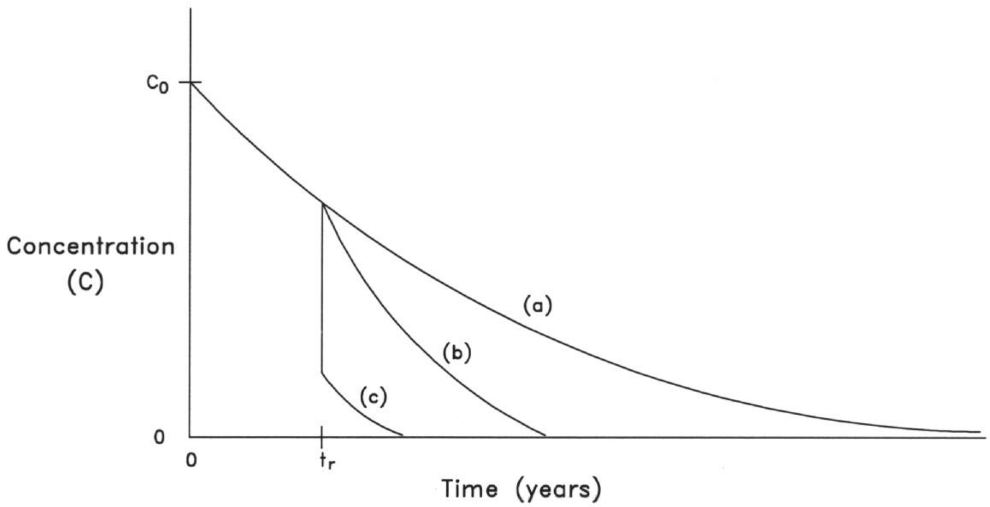

Exhibit 5.3 Concentration as a function of time of recovery: (a) = natural recovery; (b) = increased removal and/or degradation rate beginning at time t_r; (c) = bulk removal of contaminant at time of restoration t_r.

To quantify recovery with some assumed action taken (including natural recovery), loss of functionality is related to concentration as well as to the time lag in reestablishment of habitat and resource populations. In the case where the effects are small and/or sublethal, such that the habitat structure is not disrupted and recovery in the absence of toxicity would be nearly immediate, the loss of functionality is a function of concentration. The simplest model is, which may be quantifiable for a number of habitats, is that loss, L, is proportional to concentration. Below some threshold for effects, at $C = C_{min}$, $L = 0$; for $C \geq C_{max}$, $L = 1.0$, and for $C_{min} \leq C \leq C_{max}$, L increases linearly from 0.0 to 1.0 (Exhibit 5.4). The function for $C_{min} \leq C \leq C_{max}$ is:

$$L = \frac{(C - C_{min})}{(C_{max} - C_{min})} \qquad \textbf{(3)}$$

For restoration actions where all toxic concentrations are removed, there is a natural recovery curve for the reestablishment of habitat and resource populations. This recovery curve is likely sigmoidal (Exhibit 5.5) and as described by the following:

$$\frac{dPr}{dt} = P_R \, (r_b - r_b \, P_R) \qquad \textbf{(4)}$$

where P_R is the portion of full functionality at full recovery and r_b is a constant measuring rate of recovery. This function may be parameterized by estimating the time to 99% recovery (trec at L = 0.01%). Solving (analytically) the above equation for P_R, assuming $P_R = 0.01$ at $t = t_e$ (i.e., an initial condition of total loss, where t_e is the time where the habitat begins to reestablish itself) yields:

$$P_R = 1 \, / \, \left(1 + 99 \, e^{-r_b(t - t_e)}\right) \qquad \textbf{(5)}$$

The value of r_b may be estimated from an estimate of P_R at t, under conditions of no contamination and an initial condition of total loss. If $P_R = 0.99$ and $t_e = 0$, then $t = t_{rec} = 9.19/r_b$.

In the absence of toxic concentrations, replanting, restocking and other restoration actions may accelerate the recovery curve by decreasing time to recovery (Exhibit 5.5). In Chapter 3, recovery rates were estimated for various natural resources, if quantitative information is available. Recovery estimates are summarized in Section 5.2 below. Dependent natural resources might be assumed to recover proportionate to the habitat recovery, in the absence of specific information.

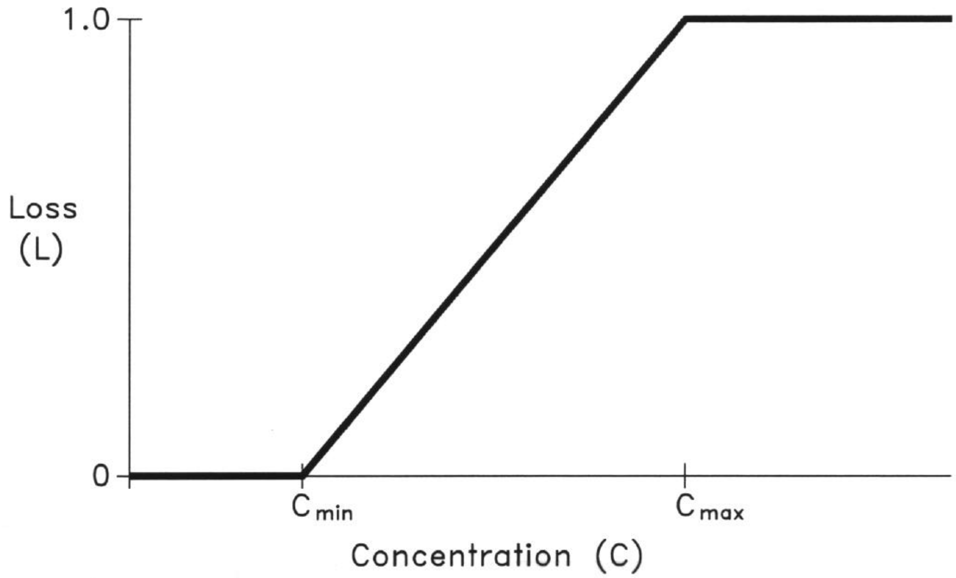

Exhibit 5.4 Hypothetical linear relationship between percent loss (L) and concentration (C).

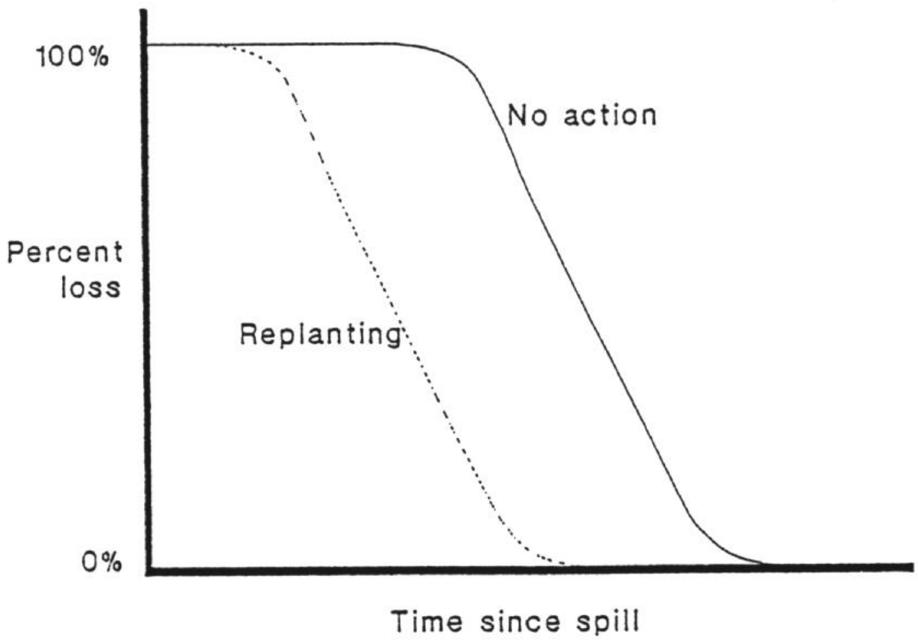

Exhibit 5.5 Case where injury is near 100% loss and restoration increases the rate of recovery.

For restoration actions where toxicity remains and reestablishment of habitat natural and populations is also required, the recovery curve needs to reflect both influences. Since remaining toxicity would inhibit the habitat and population reestablishment, the most likely model is multiplicative one of the two functions (i.e., $f_1(C) * f_2(t_e)$, where $f_1 = (1-L)$ is the function of L related to concentration (Equations 2 and 3) and f_2 is the function of P_R related to time of reestablishment (t_e) using Equation (4)). The value of t_e is the time since restoration actions were completed or since maximum concentrations were present in the case of natural recovery.

The recovery model described above could be applied to habitats, natural resources, or non-biological services (i.e., recreation), and actions for which the parameters may be quantified. The needed parameters for the simplest model are degradation rate (d), physical removal rate (r), C_{min} (threshold for effects), C_{max} (threshold for 100% loss), and time of 99% recovery (t_{rec}) or some other known percent recovery under no contamination, such that r_b may be calculated. For actions that accelerate recovery, the time to 99% recovery with the action (t_{rec}') may be used to calculate r_b' for equation (5). This yields a quantification of the portion of full recovery (P) as a function of time for the habitat or natural resource and action, where $P = P_R (1-L)$.

Various restoration actions and combinations thereof may then be compared quantitatively using these recovery curves. The analysis described in Exhibit 5.6 shows a hypothetical comparison of no action versus a selected action. The gain from the action is area B minus area A (B-A) from the exhibit, or the integrated area under each of the two curves of L = f(t). Several actions may then be compared to determine the action providing the largest gain (B-A).

Exhibit 5.7 gives some quantities for parameters for sample applications of the recovery model. Exhibit 5.8 gives resulting times to 99% recovery for the hypothetical example cases where the initial concentration is lethal to the habitat and te is taken as t at $C = C_{max}$ in Equation (2).

The cases in Exhibits 5.7 and 5.8 represent various habitat types as defined by t_{rec} (i.e., time to 99% recovery from total loss under conditions of no contamination) and k (degradation plus physical removal rate, 1/day). As can be seen in Exhibit 5.8, higher values of k speed recovery to approach that of the no contamination scenario. When k is high, restoration actions (such as doubling k or removing half of the contamination manually) performed at one year after the discharge do not have a significant effect, while they do when k is low. The hypothetical actions have much more effect if performed sooner, such as at one month. This is because of the exponential loss of concentration over time. Once concentration has fallen below C_{min}, recovery is unaffected by the removal of C or increase of k. The following gives time to $C = C_{min}$ in years for various values of k and C_o.

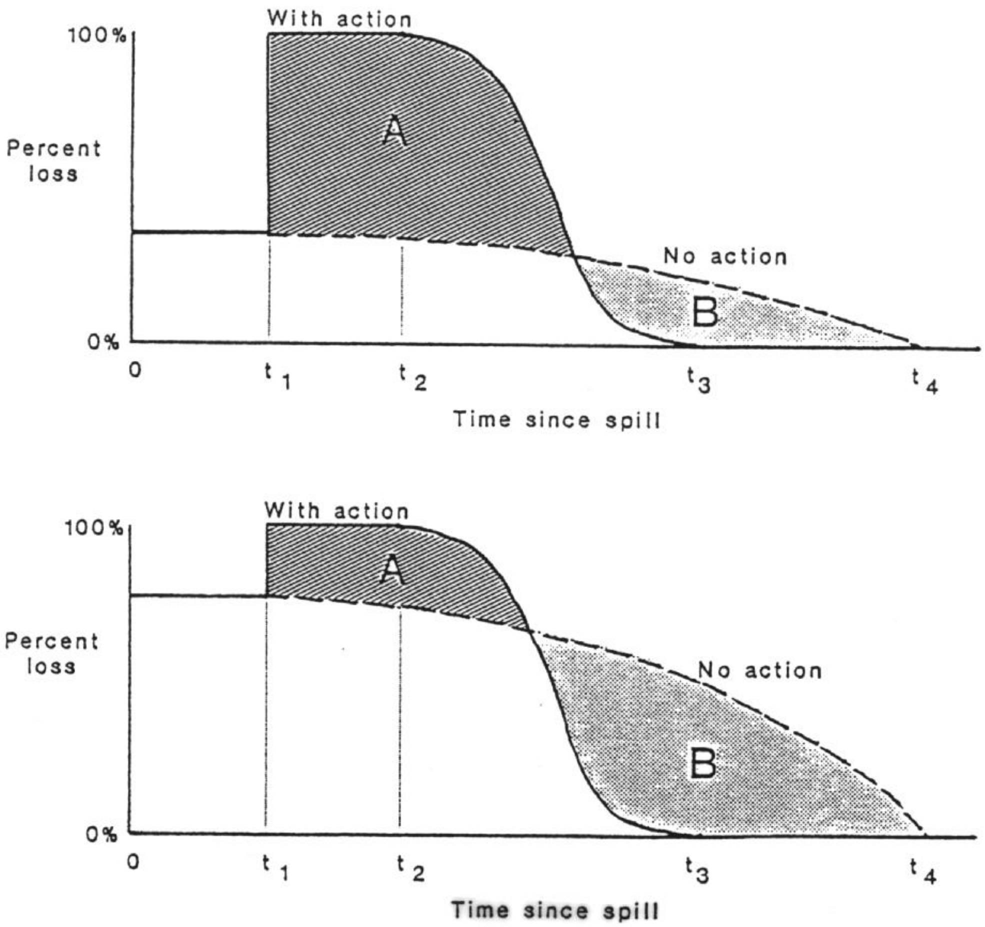

Exhibit 5.6 Schematics showing cases where restoration will or will not reduce the injury as measured by percent loss of services. Area A is the additional loss and area B is the gain resulting from performing the restoration action. The times of events are: t_1 = time work on site begins, t_2 = time work on site is completed, t_3 = time recovery is 99% complete assuming restoration action performed, t_4 = time habitat is no longer toxic assuming no restoration action is performed (i.e., assuming natural recovery). Restoration will not reduce losses if the additional loss imposed by the action is greater than the gain (upper graph, area A > area B). Restoration will reduce the injury if the gains outweigh the losses (lower graph, area A < B)

Exhibit 5.7 Estimated removal rates from Reed et al. (1989) and times to recovery when no contamination is present for several habitats.

Habitat	**r(1/day)**	**t_{rec} (yrs) (no contamination)**
Marine Exposed Rocky Shore	0.1	5
Marine Sheltered Rocky Shore	0.01	5
Marine Gravel Beach	0.005	3
Marine Sand Beach	0.01	3
Saltmarsh	0.001	15

Exhibit 5.8 Time to recovery to 99% of full function, assuming C_{min} = 0.1 ppm, C_{max} = 100.0 ppm, d=0.001/day, C_o=500 ppm, and the listed values for the parameters k and t_{rec} with no contamination.

t_{rec} (yrs) if no contamination	5	5	5	5	3	3	15
k (1/day)	0.101	0.011	0.002	0.001	0.011	0.006	0.002
Time to 99% recovery (yrs) natural recovery	5.0	5.4	8.5	16.8	3.4	3.8	17.2
Time to 99% recovery (yrs) if double k at t = 1 yr	5.0	5.4	6.6	9.0	3.4	3.7	16.6
Time to 99% recovery (yrs) if remove 50% of C at t = 1 yr	5.0	5.4	6.1	6.7	3.4	3.7	16.1
Time to 99% recovery (yrs) if double k and remove 50% of C at t = 1 yr	5.0	5.4	6.0	6.0	3.4	3.7	16.0
Time to 99% recovery (yrs) if double k at t = 1 mo	5.0	5.2	6.1	8.5	3.2	3.4	16.1
Time to 99% recovery (yrs) if remove 50% of C at t = 1 mo	5.0	5.1	5.5	6.0	3.1	3.2	15.5
Time of 99% recovery (yrs) if double k and remove 50% of C at t = 1 mo	5.0	5.1	5.3	5.5	3.1	3.2	15.3

k (1/day)	t (yrs) to $C = C_{min}$ where C_o = 500 ppm	t (yrs) to $C = C_{min}$ where C_o = 1 ppm
0.101	0.23	0.06
0.011	2.12	0.57
0.006	3.89	1.05
0.002	11.67	3.15
0.001	23.33	6.31

The above exhibit points out that improvement in recovery by removal of contamination can only be made if removal is accelerated or performed while concentrations are still above toxic thresholds. While this makes intuitive sense, it can be forgotten in practice in the urgency of trying to do something constructive. Thus, it is desirable to determine if remaining oil is indeed toxic and how long it is expected to remain at toxic concentrations. Otherwise, unnecessary and potentially harmful actions to cleanse the habitat may be unwisely undertaken.

It should also be noted that restoration actions that increase the rate of recovery ($r_{b'} > r_b$) are always beneficial to the natural resource (e.g., Exhibit 5.5). The model (equation 5 employed for the natural recovery case using r_b compared to the restoration action case of $r_{b'}$) can quantify the gain of the action.

This type of quantitative analysis allows ranking of restoration actions based on natural and restoration-enhanced recovery rates. It also allows quantification of gains for cost-benefit analyses. Such quantification can support the decisionmaking process in restoration planning.

The simple recovery model's calculations are set out as formulas for use in real situations or where the required data are available. The recovery model also serves as a construct to assist in the decisionmaking process. More sophisticated models of recovery are desirable where data may be obtained to support them. It should be noted that the available data for even the simple model may have considerable uncertainty associated with it. Probablistic modeling, sensitivity analysis, and quantification of uncertainty will elucidate risks of various actions.

5.2 Habitat Restoration and Mitigation

Exhibits 5.9 to 5.46 summarize the alternatives and actions that may be considered for habitat restoration. Restoration of a habitat includes restoration of biota and their services. Discussion of these follow.

It should again be emphasized that these are <u>actions</u> for consideration. The following discussion is not meant to be a cookbook for restoration, but to provide a basis for decision making. These exhibits point to a list of actions available for the circumstances identified. Consideration should then be made as to whether the actions will actually improve recovery under the circumstances.

5.2.1 Estuarine and Marine Wetlands

5.2.1.1 Saltmarshes

Conditions where various alternatives and actions are appropriate are summarized in Exhibits 5.9 to 5.12. Appropriate restoration actions are determined in a hierarchical fashion, depending on whether or not the oil has penetrated the substrate, is adhering to the substrate, is recoverable, the vegetation is contaminated, and vegetative (and rhizome) mortality has occurred. Actions for cases where oil has not penetrated and is not adhering to the substrate (and may or may not be recoverable) are presented in Exhibit 5.10. Exhibit 5.11 summarizes actions for cases where oil has not penetrated the substrate but is adhering to the substrate (and so is not recoverable). Exhibit 5.12 describes cases where oil has penetrated the substrate. The answers to the above 5 questions will lead the user to the available alternatives and actions for the circumstances.

Because of the potential for serious injury to saltmarsh habitats from response and restoration activities, all actions must be performed in a manner that does not result in unnecessary further injury. For example, vegetative cropping and low pressure flushing should be performed from boats in order to avoid injury to marsh substrate and vegetation root structures from trampling.

Natural recovery, vegetative cropping, low pressure flushing, replanting, and monitoring are all technically feasible. Bioremediation techniques, while potentially promising, were not tested extensively in saltmarsh habitats. Sediment removal, replacement, and replanting, along with creation, are technically feasible, but not necessarily effective or successful.

Due to the potential for serious injury, and a large body of literature documenting relatively rapid recovery on a time scale of years, natural recovery should receive first consideration in cases where oiled marshes are to be restored. If a marsh is so heavily oiled that the oil must be removed in order to prevent toxic effects on biota and/or continuing recontamination, low pressure flushing, cutting above-ground vegetation, or a combination of the two should be considered as secondary actions. Low pressure flushing can be effective if performed soon after oiling, provided oil has not penetrated the marsh substrate. If recovery does not proceed after 1-2 growing seasons, replanting should be evaluated as a tertiary action. Sediment removal and replacement should only be considered if vegetation and rhizomes are dead and the substrate is so contaminated that it impedes recovery.

The above scientific assessment does not include any technically infeasible or difficult techniques. The actions are also much less expensive than other proposed restoration actions. Thus, scientific merit (expectation of increased recovery rate) should drive the decisionmaking process for restoration of saltmarshes. Alternatives and actions are summarized in Exhibit 5.29.

It should be noted that recovery times given are based primarily on structural observations of vegetation, although data on faunal and ecological function recovery are available and influence the recovery time estimates.

5.2.1.2 Mangrove Swamp

Conditions where various alternatives and actions are appropriate are summarized in Exhibits 5.13 to 5.15. Appropriate restoration actions are determined in a hierarchical fashion, depending on whether or not the oil has penetrated the substrate, is adhering to the substrate, is recoverable, the vegetation is contaminated and plant mortality has occurred. Actions for cases where oil has not penetrated and is not adhering to the substrate are presented in Exhibit 5.14. Exhibit 5.15 summarizes actions for circumstances where oil has not penetrated but is adhering to the substrate. Exhibit 5.15 also describes circumstances where oil has penetrated the substrate.

Because of the potential for serious injury to mangrove habitats from response and restoration activities, all actions should be performed in a manner that does not result in further injury. For example, low pressure flushing should be performed from boats in order to avoid injury to the substrate, root structures, and mangrove seedlings by trampling. Cutting of vegetation and excavation of channels is unlikely to be an effective action. Such actions have resulted in increased oiling of the mangrove habitat injured in the Refineria Panama discharge (Jackson and Keller, 1991).

Natural recovery, low pressure flushing, replanting, and monitoring are all technically feasible. Bioremediation techniques have not been tested in mangrove habitats.

Natural recovery should receive primary consideration where oiled mangrove habitats are to be restored. If the environment is so heavily oiled that the oil must be removed in order to prevent toxic effects on biota and/or continuing recontamination, low pressure flushing of substrate and mangrove root systems may be performed as a secondary action, provided oil has not penetrated the substrate. If recovery does not proceed by recolonization from adjacent unoiled areas, replanting may be employed as a tertiary action. Note that sediment removal and replacement is not an effective action for mangrove restoration.

The above scientific assessment does not include any technically infeasible or difficult techniques. The actions are also much less expensive than other proposed restoration actions. Thus, scientific merit (expectation of increased recovery rate) should drive the decisionmaking process for restoration of mangrove swamps. Exhibit 5.30 summarizes alternatives and actions.

Recovery time estimates are for vegetation. Little data exist on mangrove habitat faunal recovery (except as reviewed in Section 3.2.1.2). It is assumed that fauna recovery proceeds in parallel with the vegetation.

5.2.2 Freshwater Wetlands

5.2.2.1 Emergent Wetlands

The conditions where various alternatives and actions are appropriate are the same for freshwater emergent wetlands as for saltmarshes. Thus, Exhibits 5.9 to 5.12 apply to both these habitats, as well as the discussion in Section 5.2.1.1. Exhibit 5.29 also summarizes actions in freshwater emergent wetlands, the feasibility issues, recovery rates, and costs being similar in both marsh habitats.

5.2.2.2 Scrub-Shrub Wetlands

The conditions where various alternatives and actions are appropriate are the same for all swamps, including mangrove swamps, freshwater scrub-shrub wetlands, and freshwater forested wetlands. Thus, Exhibits 5.13 to 5.15, Exhibit 5.30, and the discussion in Section 5.2.1.2 apply to this habitat as well.

5.2.2.3 Forested Wetlands

The conditions where various alternatives and actions are appropriate are the same for all swamps, including mangrove swamps, freshwater scrub-shrub wetlands, and freshwater forested wetlands. Thus, Exhibits 5.13 to 5.15, Exhibit 5.30, and the discussion in Section 5.2.1.2 apply to this habitat as well.

5.2.2.4 Bogs and Fens

Bogs and fens have developed over centuries of accumulation of peat and require extremely long recovery times (decades to centuries) following any alteration or removal of the substrate. For this reason, the only recommended alternatives and actions are natural recovery and bioremediation (Exhibit 5.16). The latter remains untested, but may be helpful to speed degradation of oil contamination. Costs for this action are unknown, but presumably similar to those for saltmarshes and emergent wetlands (Exhibit 5.29).

5.2.3 Vegetated Beds

5.2.3.1 Macroalgal Beds (Estuarine and Marine)

5.2.3.1.1 Intertidal Macroalgal Bed

The important elements of intertidal macroalgal bed restoration are, to a large extent, coincident with those for the rocky intertidal area. To the extent that the intertidal macroalgal bed is unique, it is considered in Exhibits 5.17 and 5.34. Careful cleanup (in both the response and restoration context) to avoid aggravating injuries is called for. Vegetative cropping may be needed if oil adheres to the vegetation. While replanting is proposed as a potential action, it remains untested as viable.

5.2.3.1.2 Kelp Bed

Alternatives and actions for kelp bed restoration are summarized in Exhibit 5.18 and Exhibit 5.35. Contaminated vegetation may be cropped. In most cases it is expected that natural recovery will be the action of choice. The time to full community recovery is uncertain because the faunal response to oil discharges is largely unknown. Replanting methods exist but have not been used in restoring oil discharge injuries. Herbivore control might be needed during the period of restoration to accelerate recovery.

5.2.3.2 Seagrass Beds

Seagrasses do not appear to be especially sensitive to oil discharges but their faunal communities may be quite sensitive. Restoration actions for seagrass beds are summarized in Exhibits 5.19 and 5.36. It is important to note that maintaining the integrity of the sediment may be important to restoration efforts whether or not replanting is attempted (Zieman et al., 1984). Also, off-site restoration, if chosen, should only be attempted in areas where seagrass is known to grow (e.g., a degraded seagrass bed in an areas where the cause of degradation is believed to have abated) (Zieman and Zieman, 1989). As with other complex habitats, the time to recovery for the plants can be projected. However, there exists only a vague idea of how rapidly the community is restored to full function. It is generally assumed that a structurally-restored grass bed will recolonize with its typical fauna from surrounding uninjured areas.

5.2.3.3 Freshwater Aquatic Bed (Submerged and Floating Vegetation)

There is little information on recovery of freshwater aquatic beds from oil discharge impacts. These habitats are not always considered valued so much as a nuisance. Possible restoration actions are summarized in Exhibit 5.20 and 5.37. Some of the information in these exhibits, such as restoration time, are speculative in the absence of more data.

5.2.4 Mollusc (Oyster) Reefs

Alternatives and actions for oyster reef restoration are summarized in Exhibits 5.21 and 5.38. There is no available information on restoration of oyster beds in response to oil discharge injuries. If oysters survive the discharge, they may still require some period of depuration before they are useful as a fishery resource. Natural reseeding may be quite rapid in some places at certain times of year, but will have to be augmented under other conditions. Where the oyster bed is heavily injured through response efforts, reconstruction and reseeding may be appropriate.

5.2.5 Coral Reefs

Restoration alternatives and actions for coral reefs are summarized in Exhibit 5.22 and Exhibit 5.39. This information is based on a rather sparse history of coral reef recovery from oil discharge injury. Because coral is so slow-growing, it is reasonable to assume that when the reef has recovered, the community has recovered. Unfortunately, there is little data to support this supposition.

5.2.6 Estuarine and Marine Intertidal

5.2.6.1 Rocky Shores

Conditions where various alternatives and actions are appropriate in the restoration of rocky shores is outlined in Exhibit 5.23. The actions are for oil-affected estuarine, marine, and freshwater rocky (and artificial) shores. Exhibit 5.40 further describes these alternatives and actions, including restrictions to be effective, feasibility, recovery times, and costs. Restoration actions are determined based on the importance of biological verses non-biological services, whether oil has adhered to the surface, and access to the shoreline. Where biological services of the rocky shore are the primary concern, only natural recovery and possibly bioremediation are recommended. Non-biological services will be more important in certain recreational-use areas, harbors, and other high-use areas. The value of these non-biological services may justify such extreme measures (in terms of biological effects) as hot water washing and sandblasting. Concerns over contamination of nearby habitats and biota may justify more rigorous cleaning as well.

5.2.6.2 Cobble-Gravel Beaches

Restoration alternatives and actions for cobble-gravel beaches are outlined in Exhibit 5.24. The decision for choosing an action is determined first by the importance of biological verses non-biological services. When biological services are important, bioremediation may be considered in low energy environments. However, natural recovery should be the preferred alternative in high energy areas where fertilizers would not remain on the shoreline to be effective. Where non-biological services are important, or where contamination to other natural resources is a concern, the decision on restoration is determined by whether non-biological services or other natural resources should take precedence, and whether or not oil has penetrated the substrate. Exhibit 5.41 summarizes the possible actions. Sediment agitation includes berm relocation and sediment mixing.

5.2.6.3 Sand Beaches

Estuarine, marine, and freshwater sand beaches injured by oil may be restored by the actions outlined in Exhibit 5.25. Again, actions are determined by the importance of biological versus non-biological services, concerns for contamination of other nearby resources, the energy of the environment, and penetration of oil into the substrate. Actions are reviewed in Exhibit 5.42.

5.2.6.4 Intertidal Mud Flat

Exhibit 5.26 outlines the appropriate actions for restoration of marine, and estuarine intertidal mud flats, and freshwater silt-mud shores. This is also reviewed in Exhibit 5.43. Alternative actions depend on penetration of the oil into the substrate and the toxicity of contaminated sediment.

5.2.7 Estuarine and Marine Subtidal

5.2.7.1 Subtidal Rock Bottoms

Natural recovery is the only alternative for restoration of estuarine, marine, and freshwater rock bottoms (Exhibit 5.44).

5.2.7.2 Subtidal Cobble-Gravel, Sand, and Silt-Mud Bottoms

Estuarine and marine subtidal cobble-gravel, sand and silt-mud bottoms, and freshwater sand and silt-mud bottoms restoration actions are outlined in Exhibit 5.27. The appropriate action is determined by whether or not oil has penetrated the substrate and is at toxic concentrations. If not, natural recovery is likely preferable. If the sediment is toxic, removal or capping may be used depending on the physical characteristics of the discharge area. Alternatives and actions are summarized in Exhibit 5.45.

5.2.8 Riverine and Lacustrine Shorelines

5.2.8.1 Rock Shores

Freshwater rock shores would be treated the same as estuarine and marine rock shore. (See Section 5.2.6.1.)

5.2.8.2 Cobble-Gravel Shores

Freshwater cobble-gravel beaches would be treated the same as estuarine and marine cobble-gravel shore. (See Section 5.2.6.2.)

5.2.8.3 Sand Shores

Freshwater sand shores would be treated the same as estuarine and marine sand beaches. (See Section 5.2.6.3.)

5.2.8.4 Silt-Mud Shores

Freshwater silt-mud shores would be treated the same as estuarine and marine intertidal mud flats. (See Section 5.2.6.4.)

5.2.9 Riverine and Lacustrine Unvegetated Bottom

5.2.9.1 Rock Bottom

Natural recovery is the only alternative for restoration of estuarine, marine, and freshwater rock bottoms (Exhibit 5.44).

5.2.9.2 Cobble-Gravel Bottom

Exhibit 5.28 outlines available restoration actions for freshwater cobble-gravel bottoms. Where oil is adhering to or within the substrate, dredging and replacement may be considered. Streambed agitation is an action in riverine habitats. Exhibit 5.46 reviews these actions.

5.2.9.3 Sand and Silt-Mud Bottom

Freshwater sand and silt-mud bottoms would be treated the same as estuarine and marine subtidal bottoms of the same substrate type. (See Section 5.2.7.2)

5.2.10 Monitoring of Habitat Recovery

Monitoring costs have been estimated for a generic monitoring plan on a unit basis, the unit being an individual stratum or area of uniform habitat and environmental conditions. The description of the stratum is in Section 3.2.10. It is not a cost per unit area (such as $/ha), but rather a cost per stratum of affected habitat. Thus, Exhibits 5.29 to 5.46 contain the symbol M to refer to monitoring costs.

The value of M, monitoring cost per stratum, is estimated and described in Section 4.4. The costs of sediment monitoring (Section 4.4) are relevant to most habitat monitoring (M). Thus, the value of M would be on the order of $5,000 to $125,000 per year (1992$), depending on the complexity of the sampling and testing required.

5.3 Biological Populations

Alternatives and actions for biological resource populations may be summarized by the following:

- Natural recovery monitoring;
- Harvest alternation;
- Harvest refugia;
- Stocking, culturing, and seeding;
- Relocation;
- Habitat enhancement;
- Artificial structures;
- Facilitation of migration;
- Habitat protection and acquisition; and
- Replacement of services.

The specific actions are very species- and site-specific, and, therefore, cannot be summarized as concisely as for habitats in the previous section.

Factors that may need to be considered in developing and evaluating alternatives and actions include:

- Objectives should be carefully laid out and specific to the target species, life history requirements, and prevailing environmental conditions;

- Effectiveness and success should be rigorously evaluated. One should not assume that doing something has benefit. This has often been the case historically;

- The desire to solve waste disposal and other needs should not be considered a mitigating factor for restoration of injured natural resources unless proven to be truly effective at restoring those natural resources or services injured;

- Where possible, estimated costs should be weighed against restoration benefits;

- Attention should be paid to impacts on non-target species. The net benefits to all natural resources must be evaluated as a whole;

- In considering stocking efforts, the maintenance of genetic integrity in a wild stock is crucial. Also, possible introduction of disease should be considered;

- Enhancement actions may prove more effective than direct restoration of oil-injured natural resources because of lack of effectiveness of the latter;

- Changes in management practices resulting in benefits to both natural resources and their services is a preferred action.

- Restoration of habitats chronically affected by toxins and water quality problems or development can effectively replace oil-injured natural resources if replacement stocks are reduced but still viable; and

- Monitoring of injuries and recovery is crucial but may be difficult due to natural variability. Adequate financial resources must be applied to this part of the restoration effort to ensure the success of the restoration.

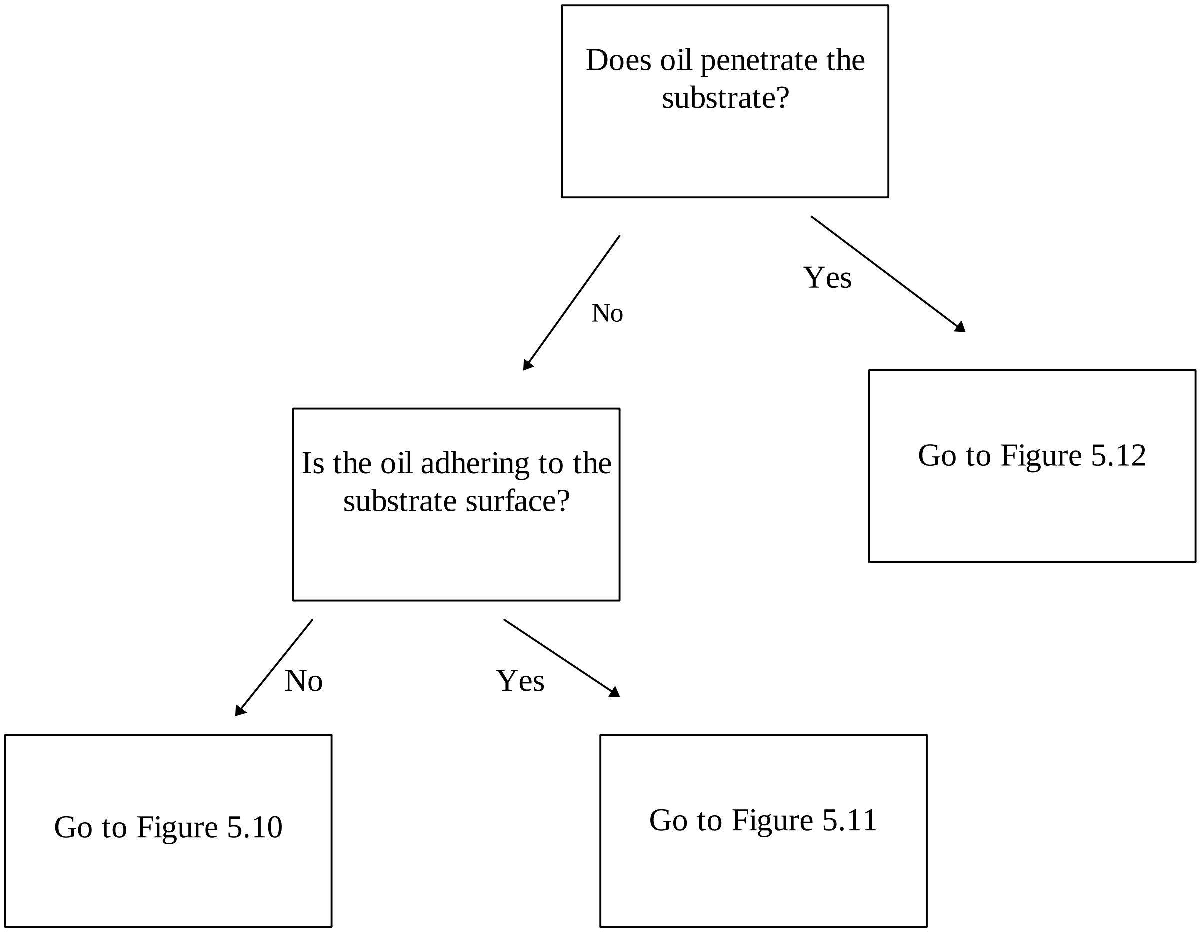

Exhibit 5.9 Decision diagram for restoration alternatives and actions for saltmarsh and freshwater emergent wetlands.

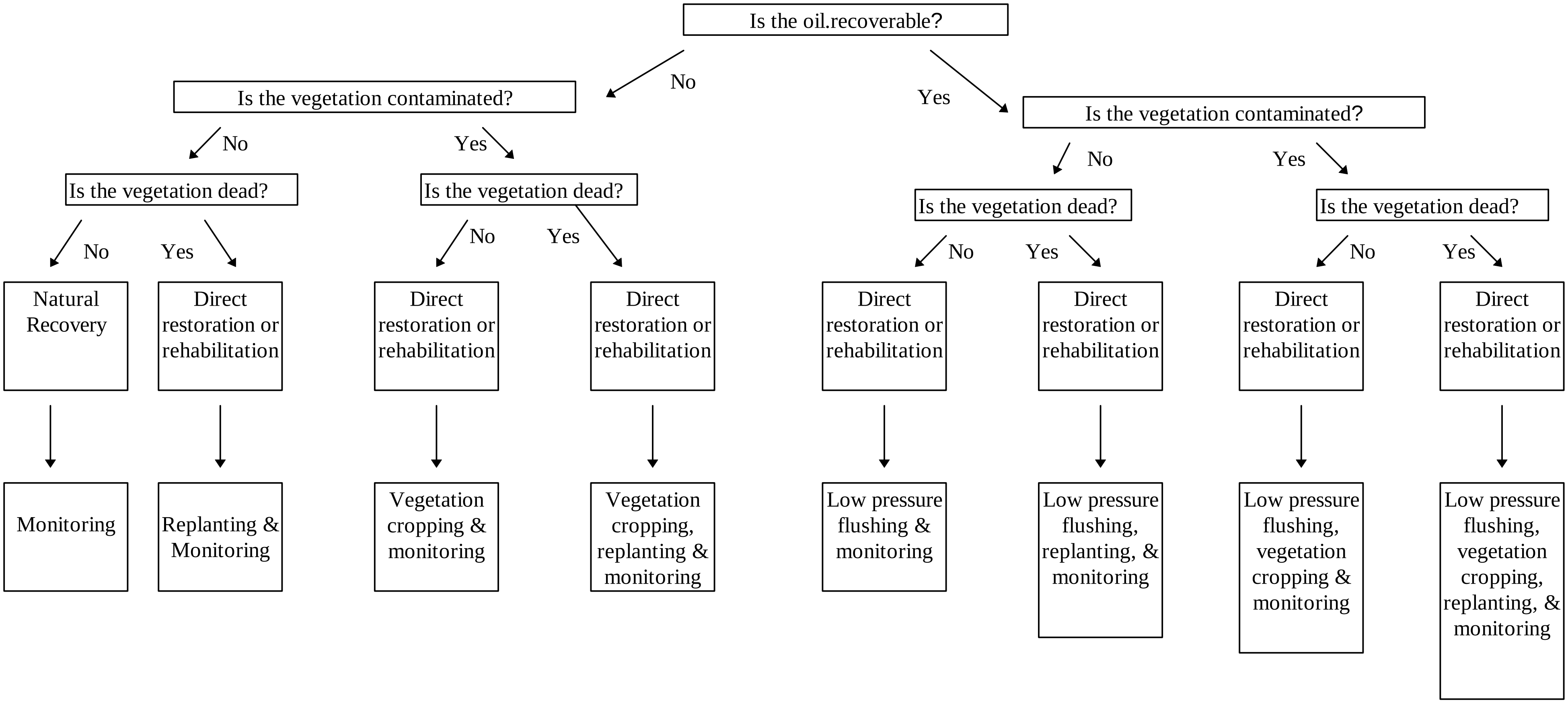

Exhibit 5.10 Decision diagram for restoration alternatives and actions for saltmarsh and freshwater emergent wetlands where oil has not penetrated the substrate and is not adhering to the substrate.

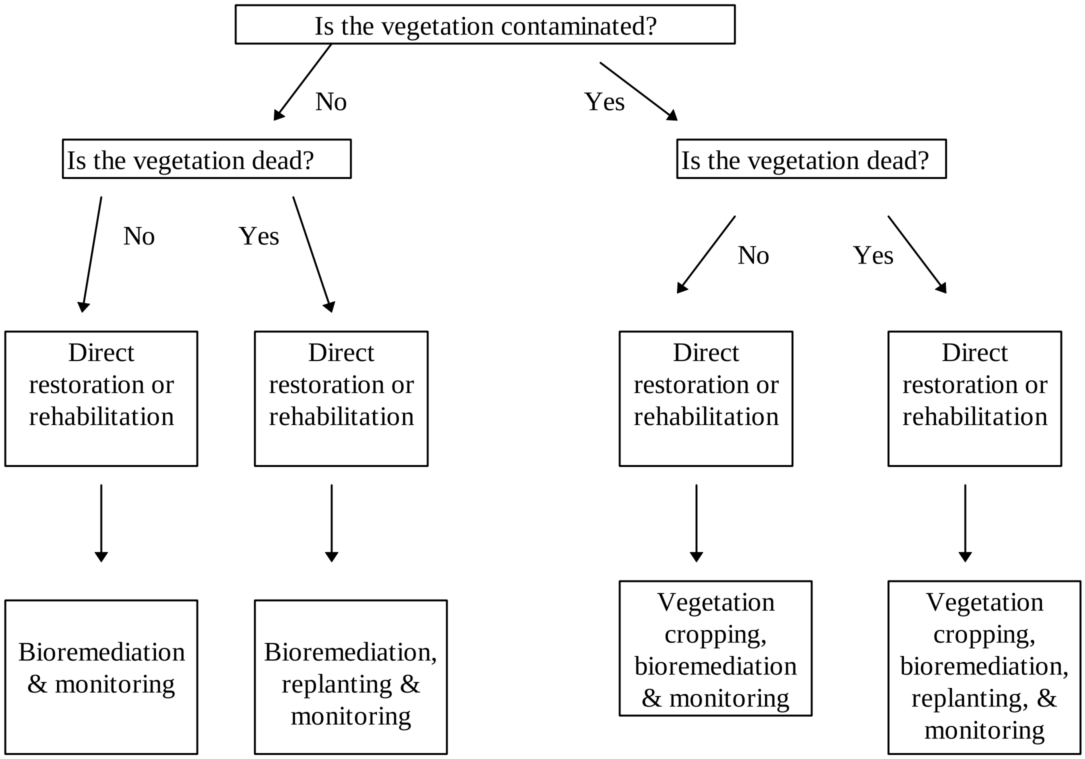

Exhibit 5.11 Decision diagram for restoration alternatives and actions for saltmarsh and freshwater emergent wetlands where oil has not penetrated but is adhering to the substrate (and so is not recoverable).

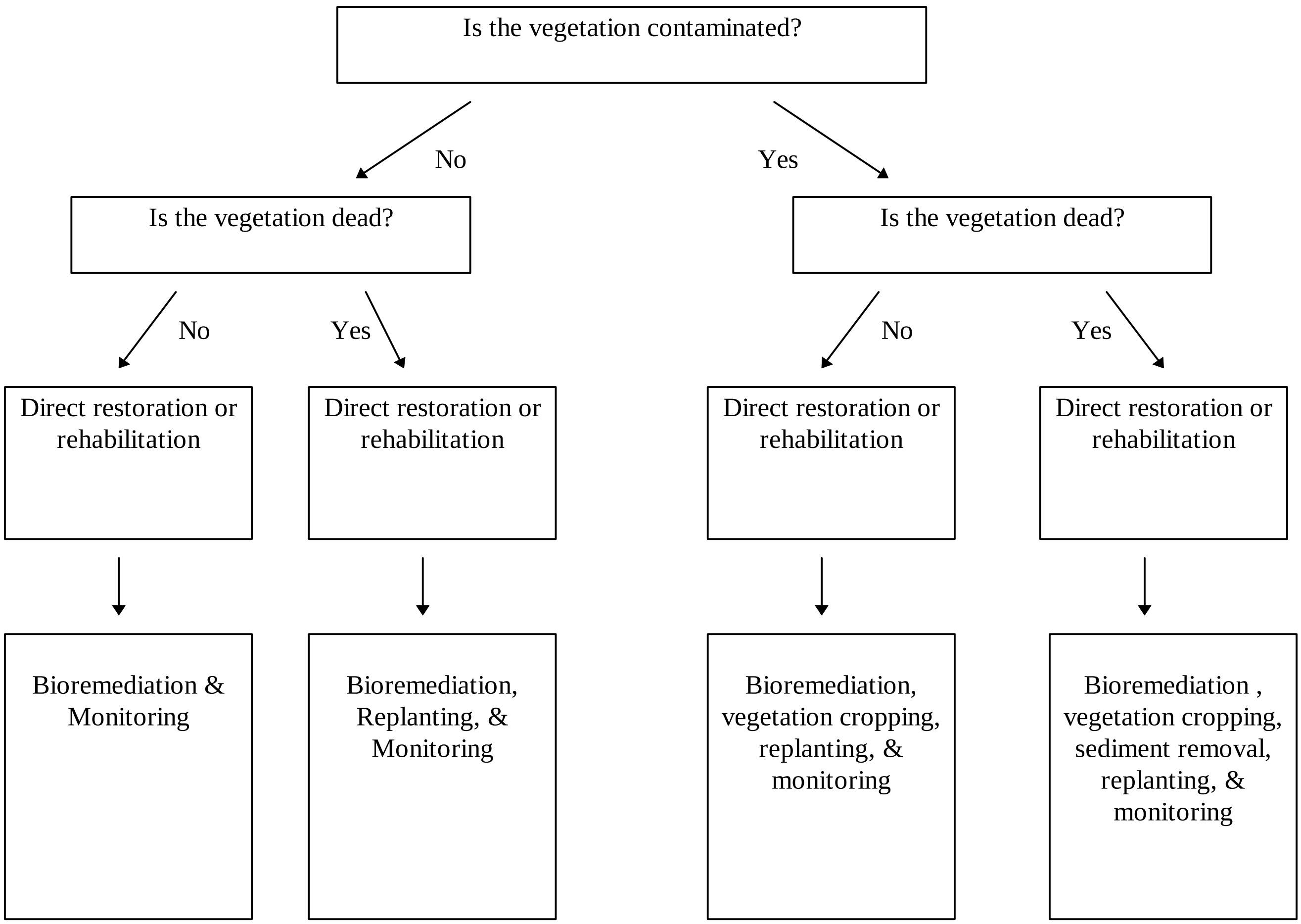

Exhibit 5.12 Decision diagram for restoration alternatives and actions for saltmarsh and freshwater emergent wetlands where oil has penetrated the substrate.

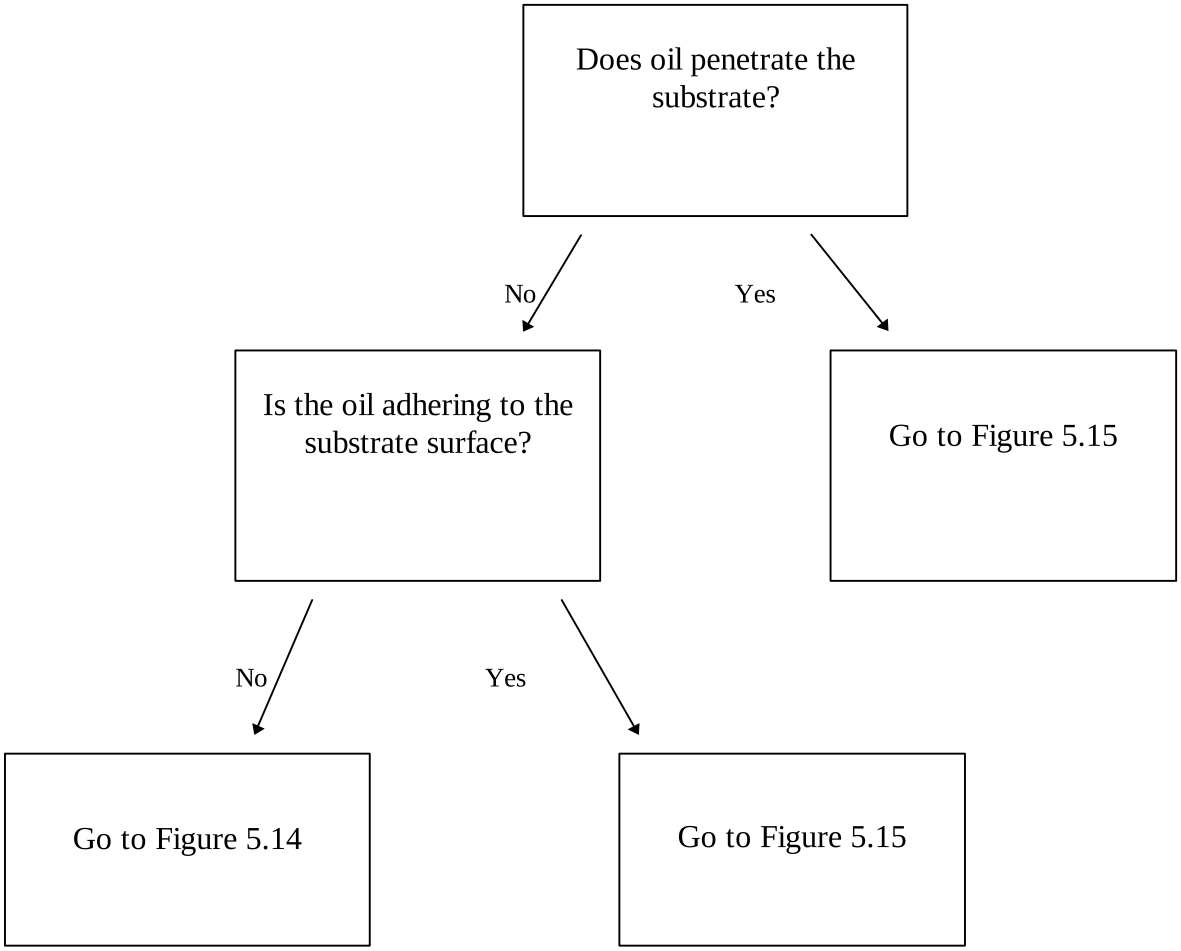

Exhibit 5.13 Decision diagram for the restoration alternatives and actions for mangrove swamps, freshwater scrub-shrub wetlands, and freshwater forested wetlands.

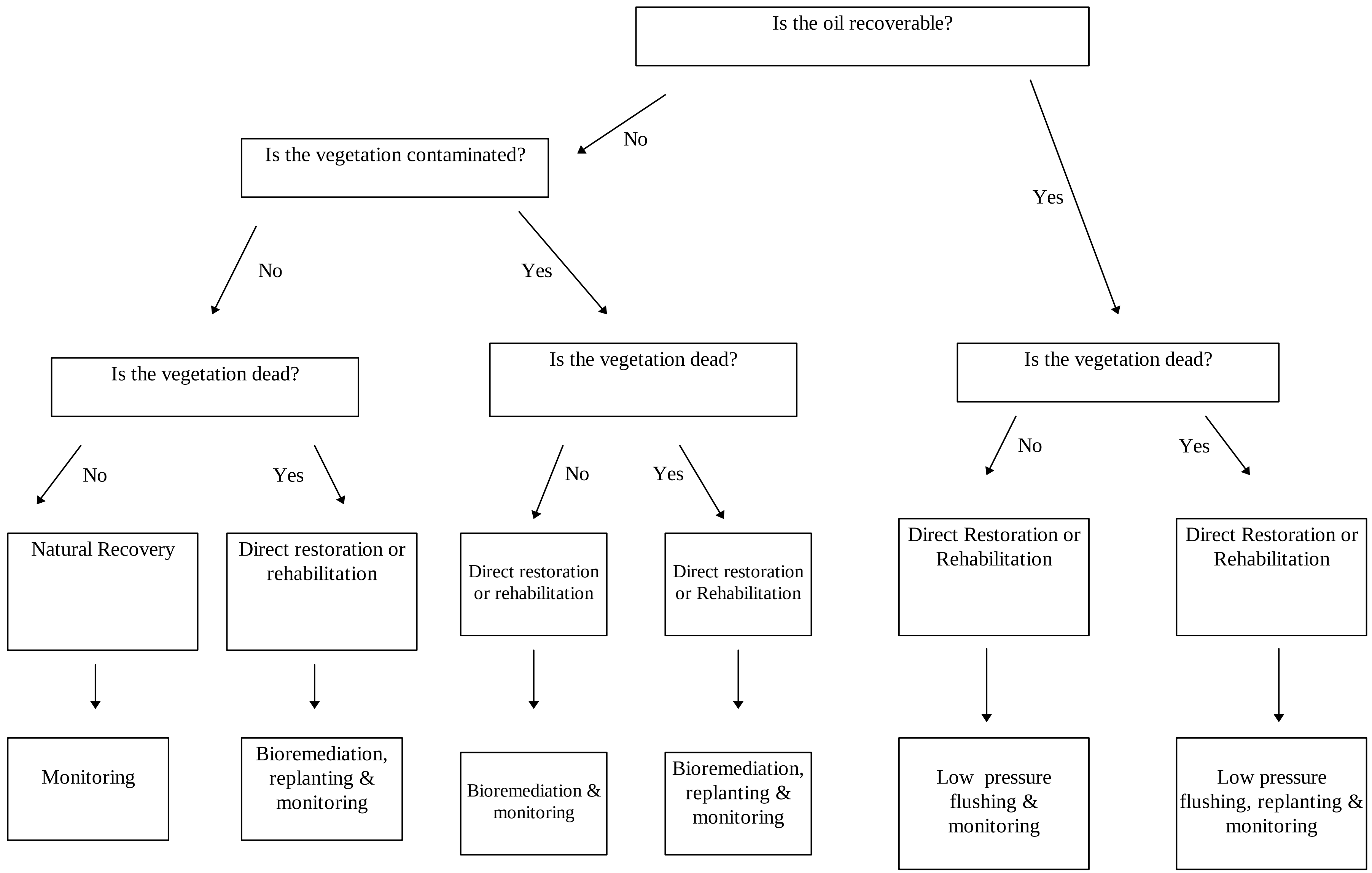

Exhibit 5.14 Decision diagram for restoration alternatives and actions for mangrove swamps, freshwater scrub-shrub wetlands, and freshwater forested wetlands where oil has not penetrated the substrate and is not adhering to the substrate.

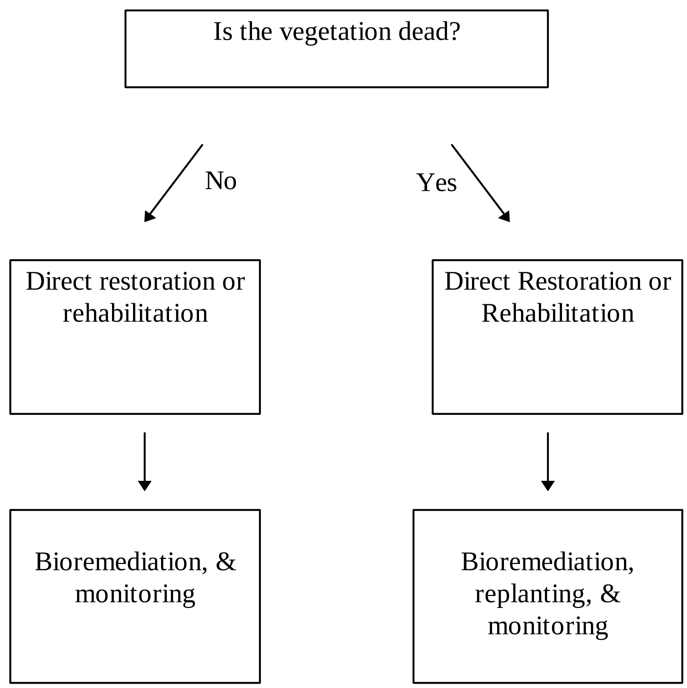

Exhibit 5.15 Decision diagram for restoration alternatives and actions for mangrove swamps, freshwater scrub-shrub wetlands, freshwater forested wetlands where oil may or may not have penetrated but is adhering to the substrate.

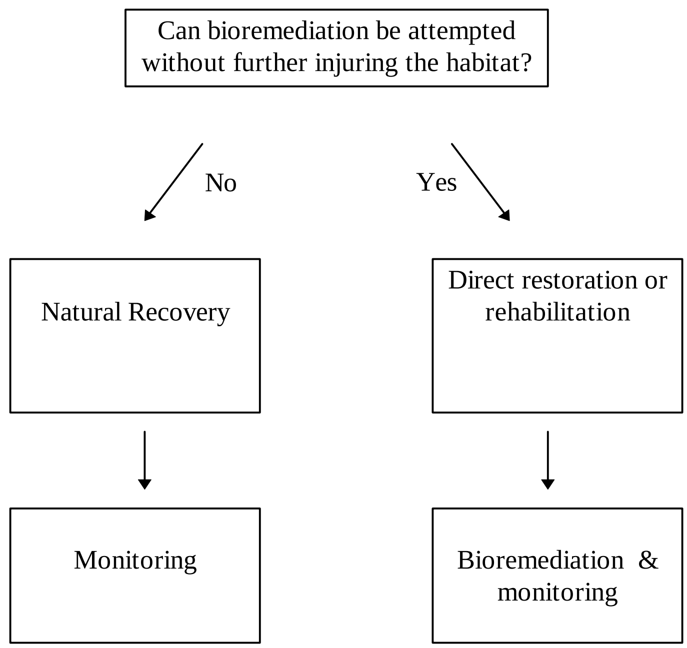

Exhibit 5.16 Decision diagram for restoration alternatives and actions for freshwater bogs and fens.

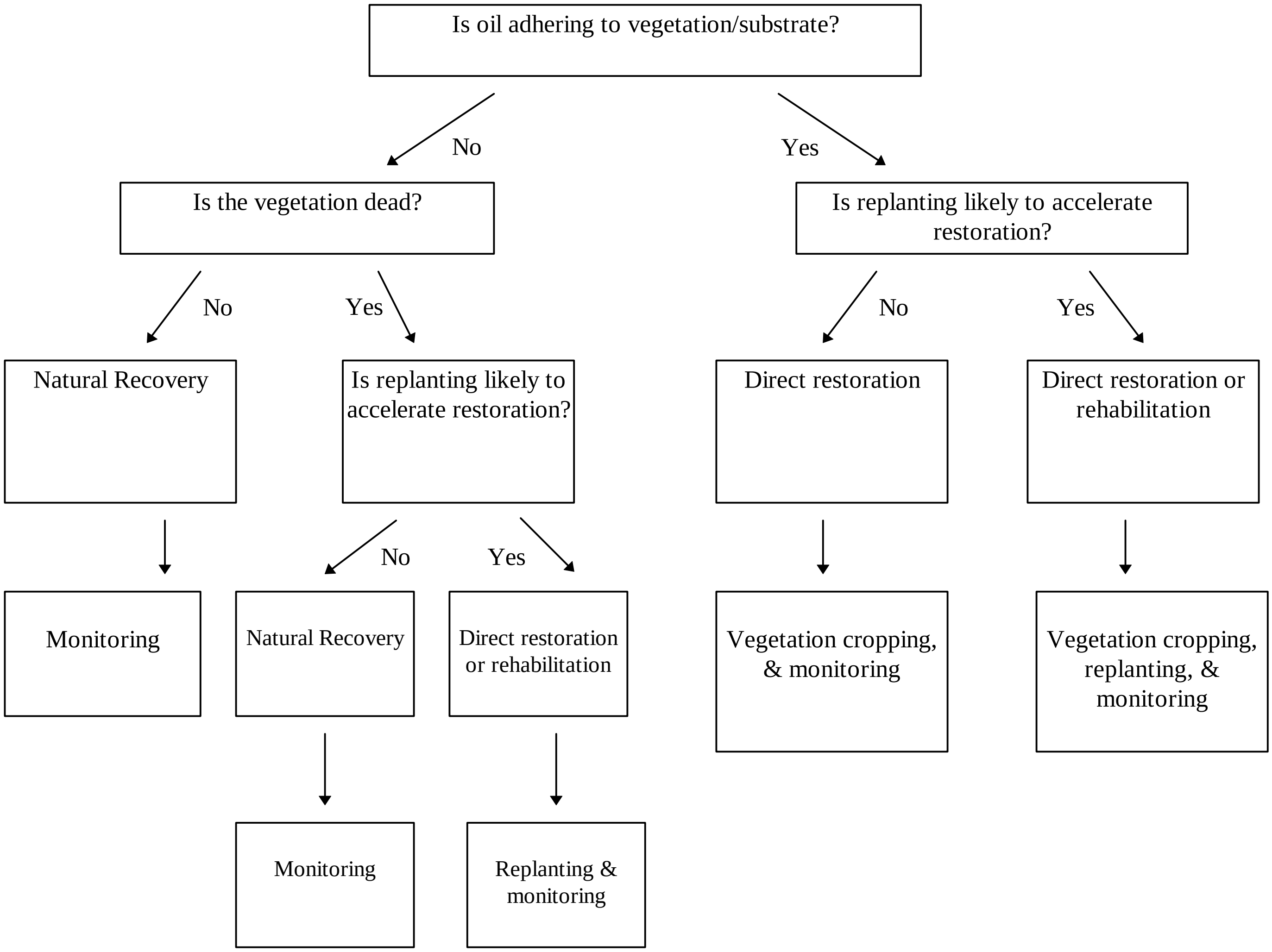

Exhibit 5.17 Decision diagram for restoration alternatives and actions for internal macroalgal beds.

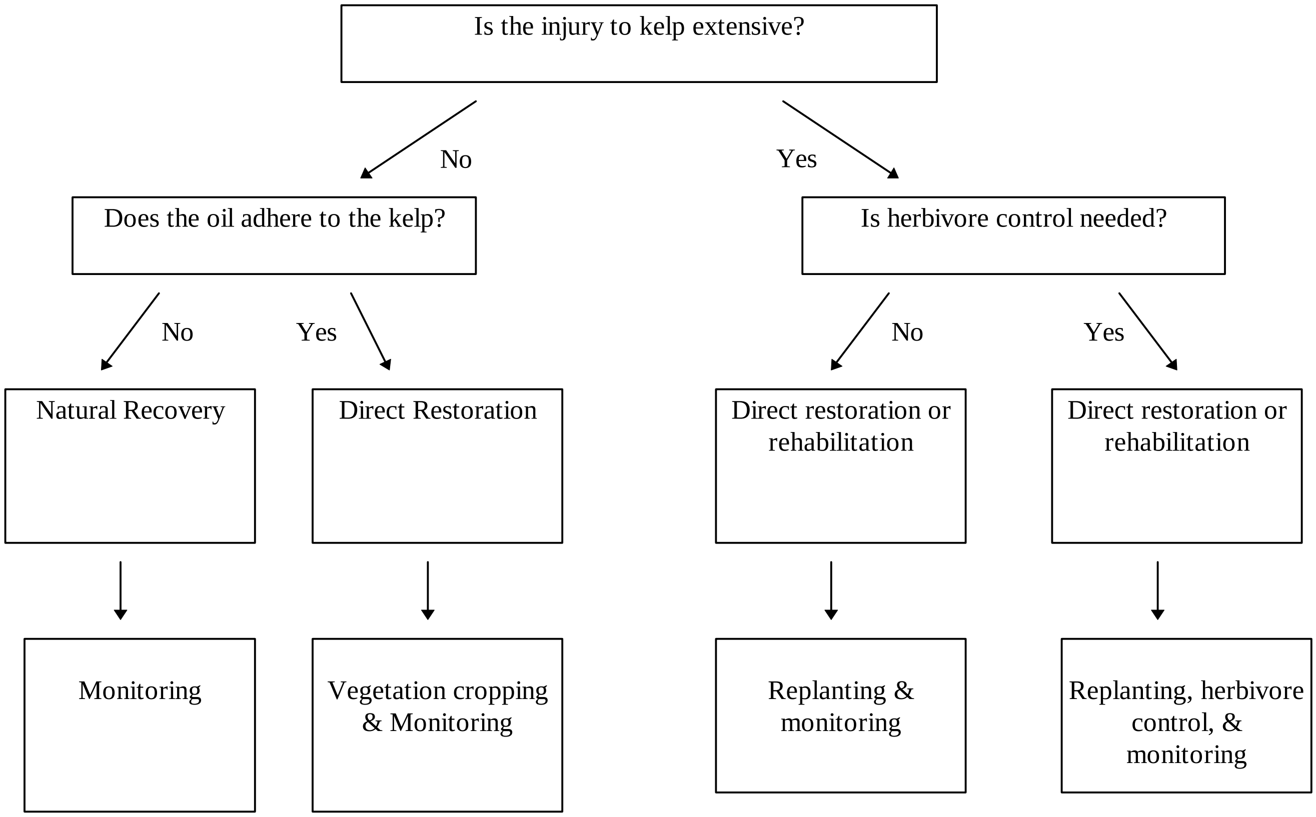

Exhibit 5.18 Decision diagram for restoration alternatives and actions for kelp beds.

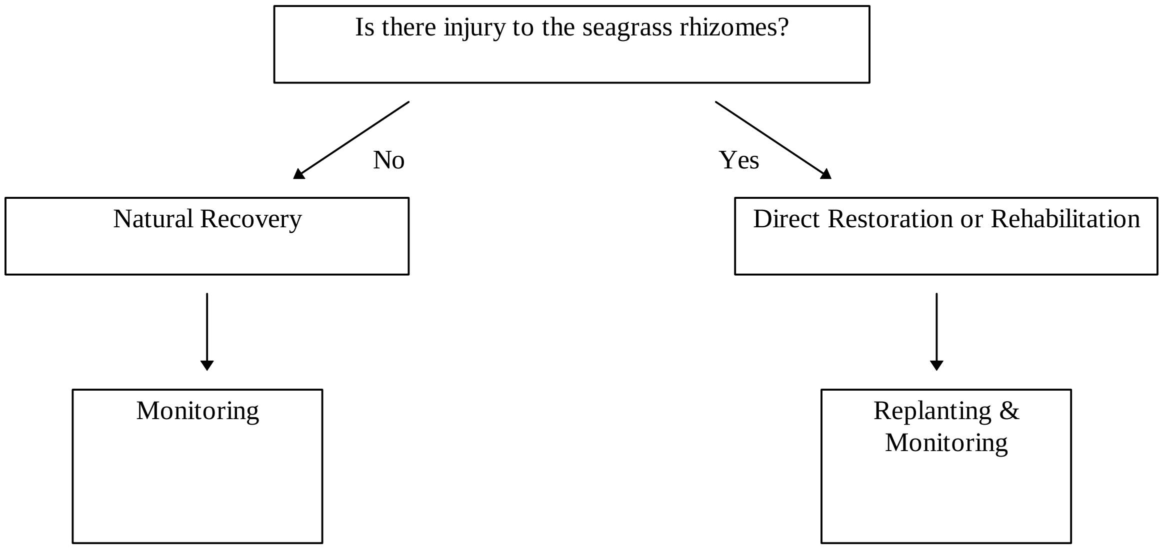

Exhibit 5.19 Decision diagram for restoration alternatives and actions for seagrass beds.

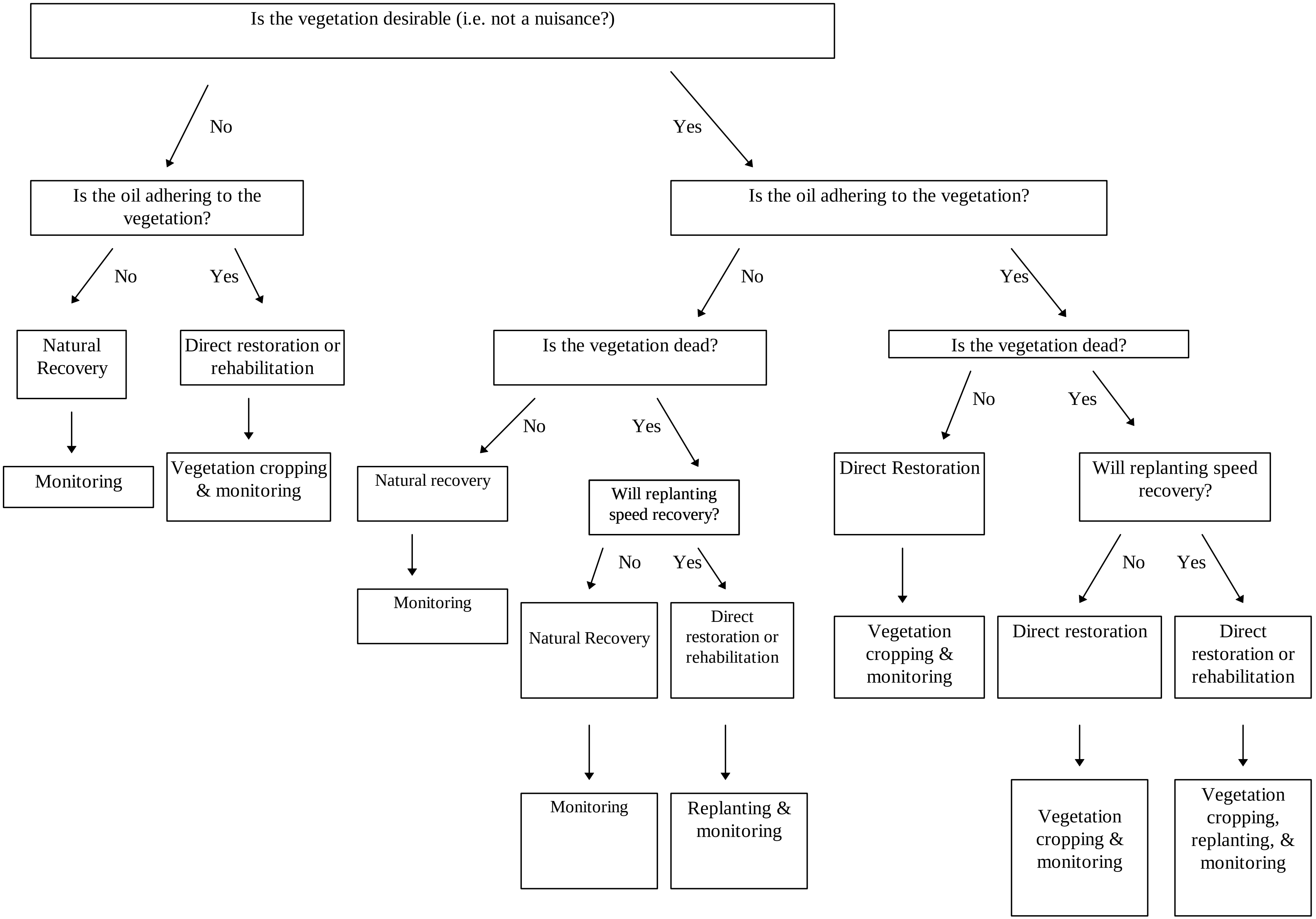

Exhibit 5.20 Decision diagram for restoration alternatives and actions for freshwater aquatic beds.

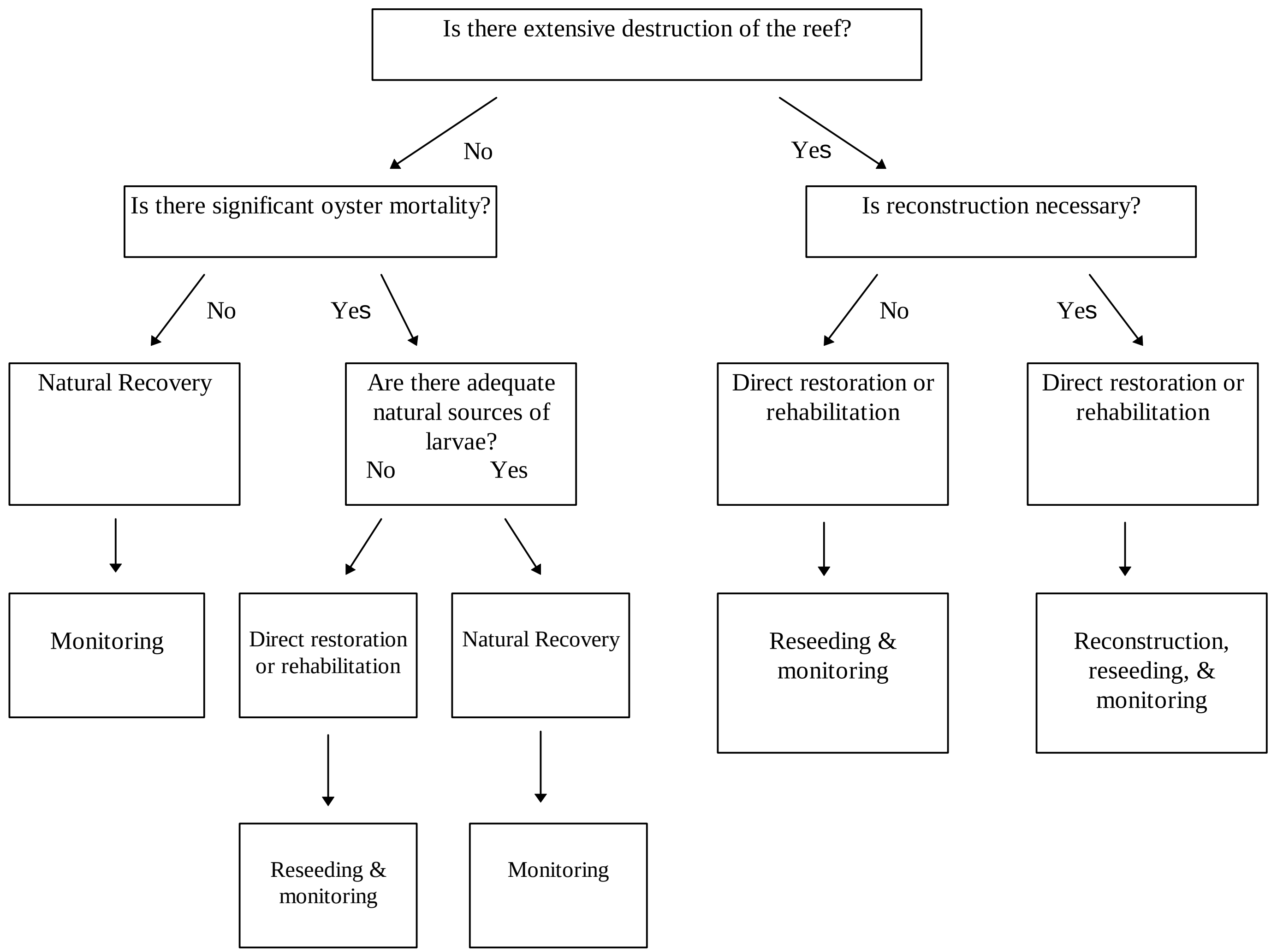

Exhibit 5.21 Decision diagram for restoration alternatives and actions for oyster reefs.

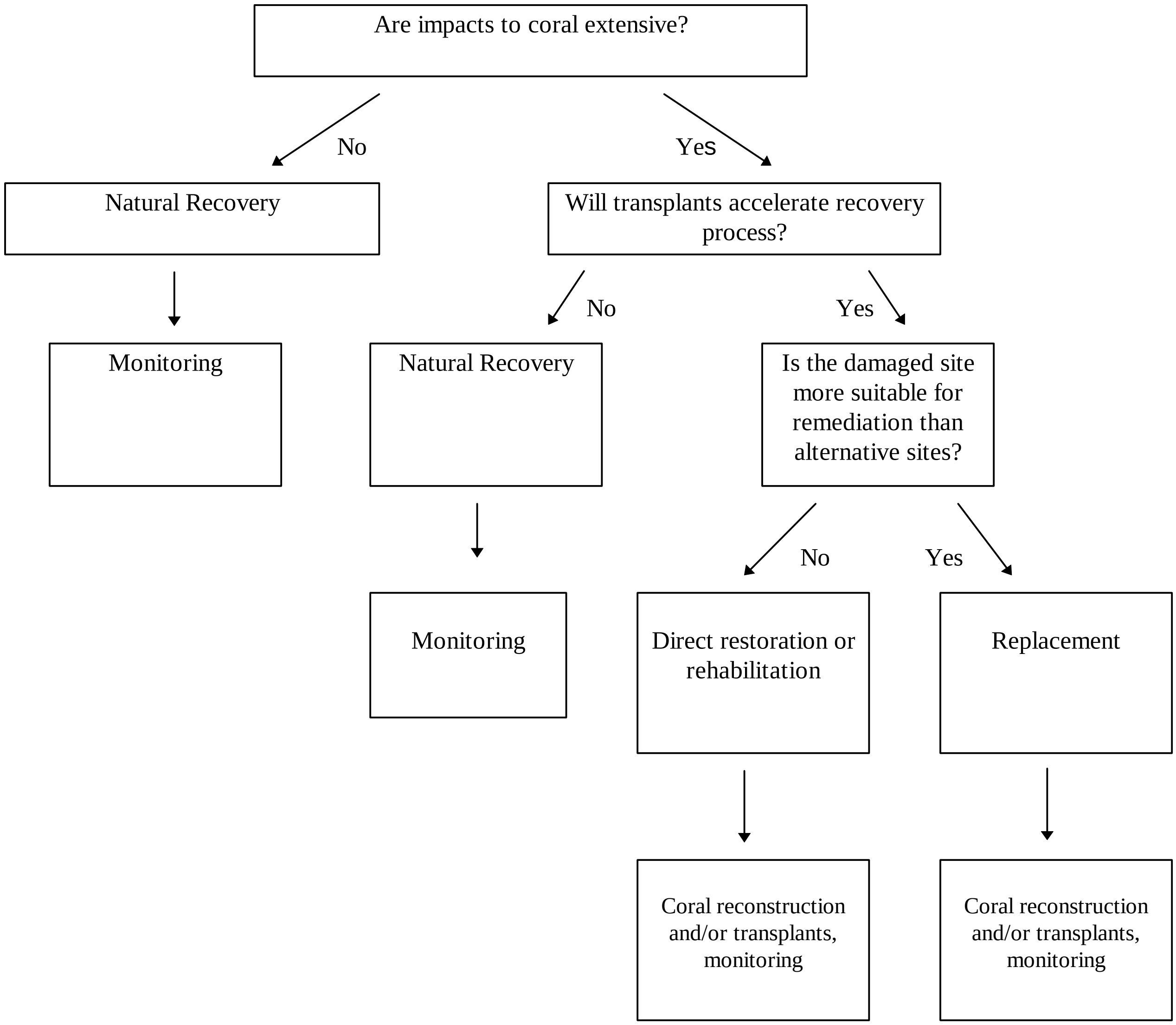

Exhibit 5.22 Decision diagram for restoration alternatives and actions for coral reefs.

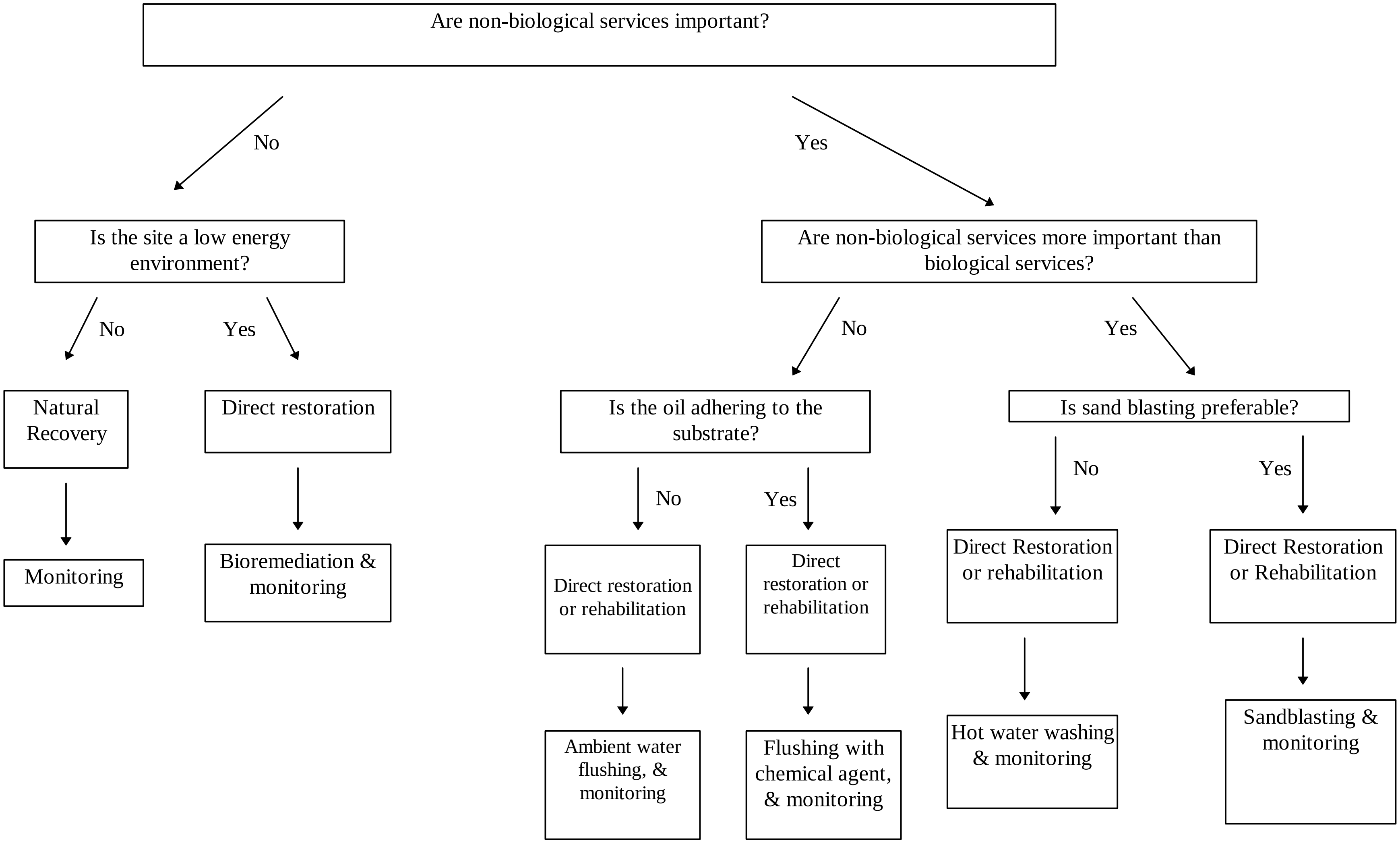

Exhibit 5.23 Decision diagram for restoration alternatives and actions for estuarine, marine and freshwater rocky shores.

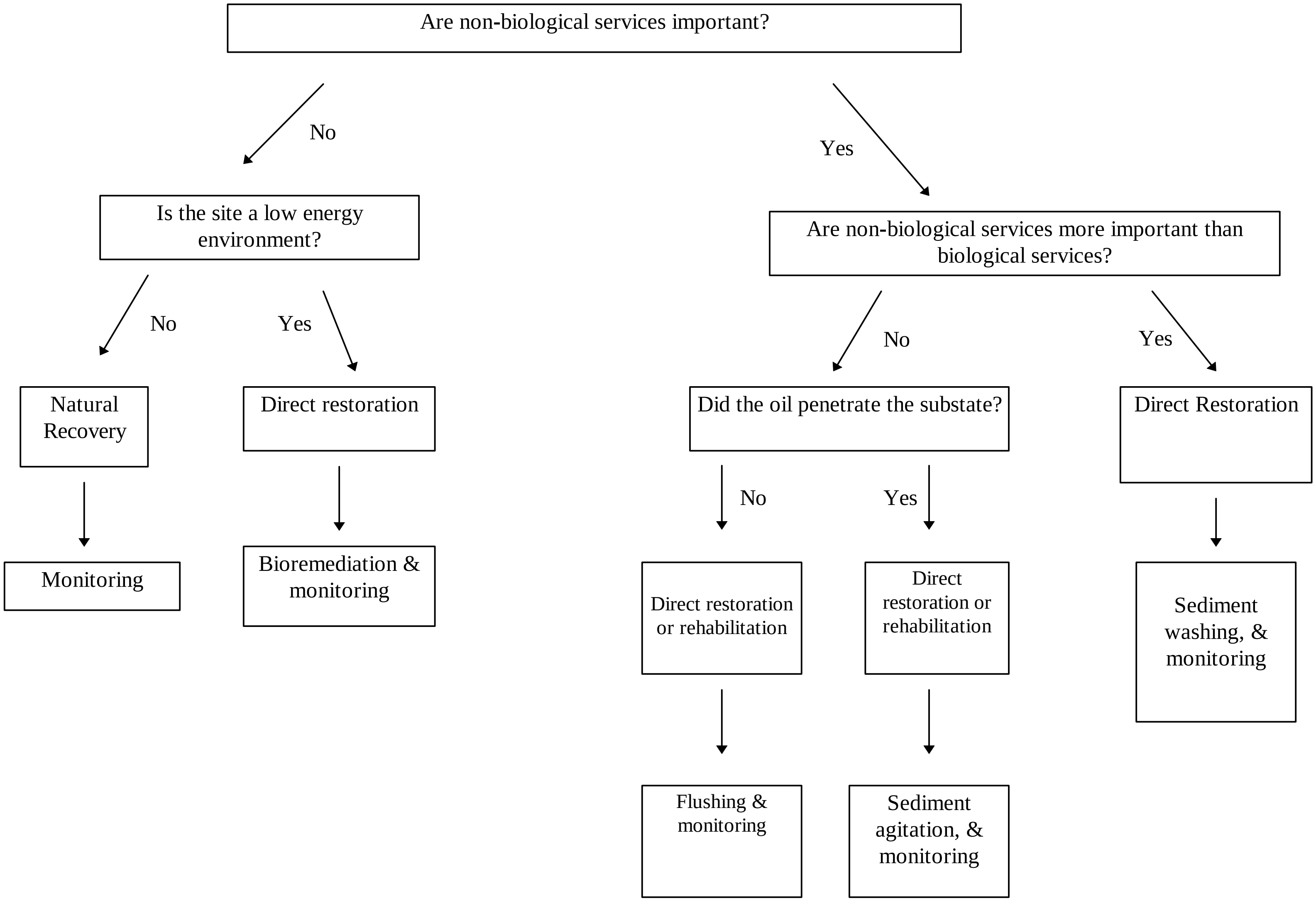

Exhibit 5.24 Decision diagram for the restoration alternatives and actions for estuarine, marine and freshwater cobble-gravel beaches.

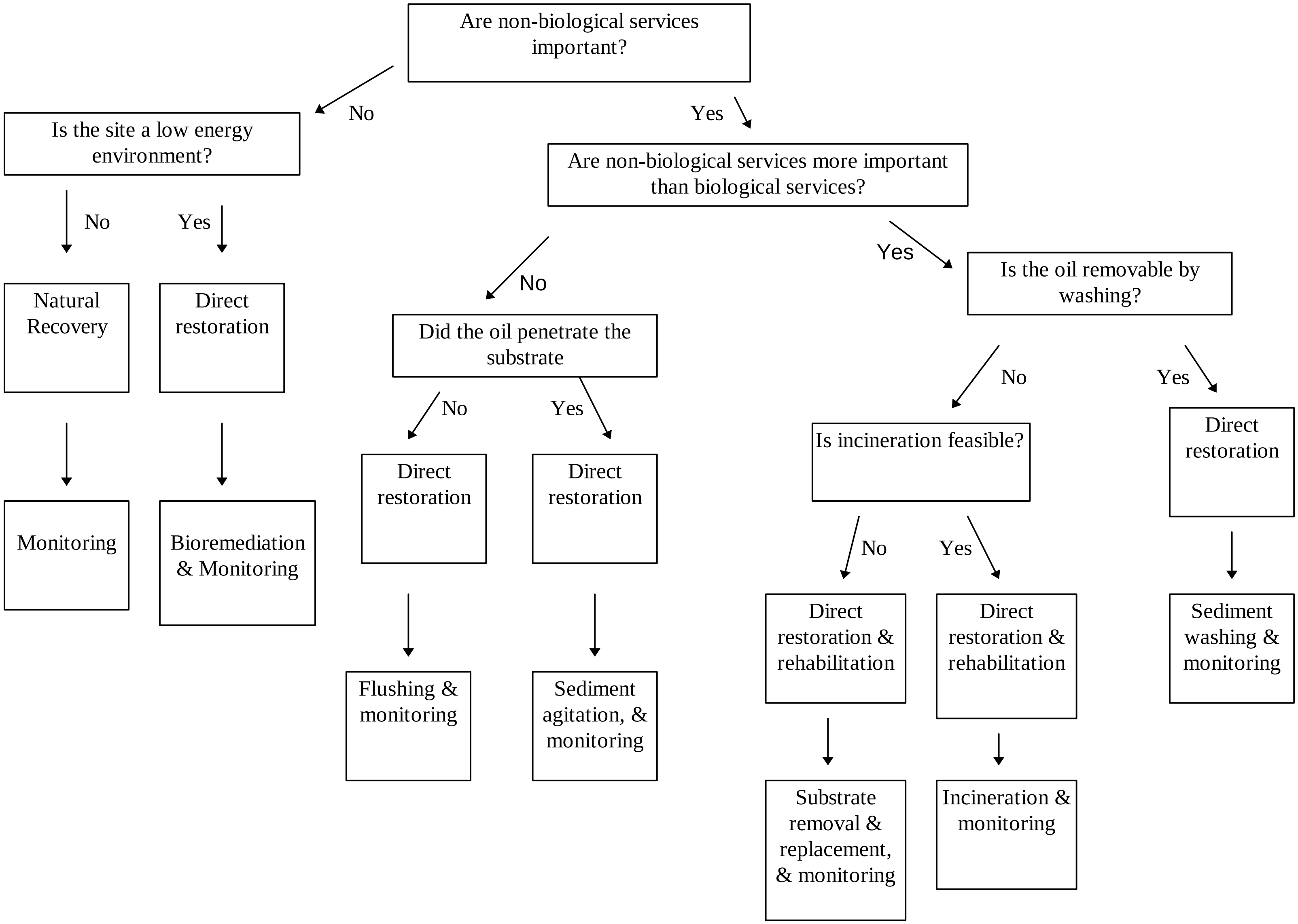

Exhibit 5.25 Decision diagram for restoration alternatives and actions for estuarine, marine and freshwater sand beaches

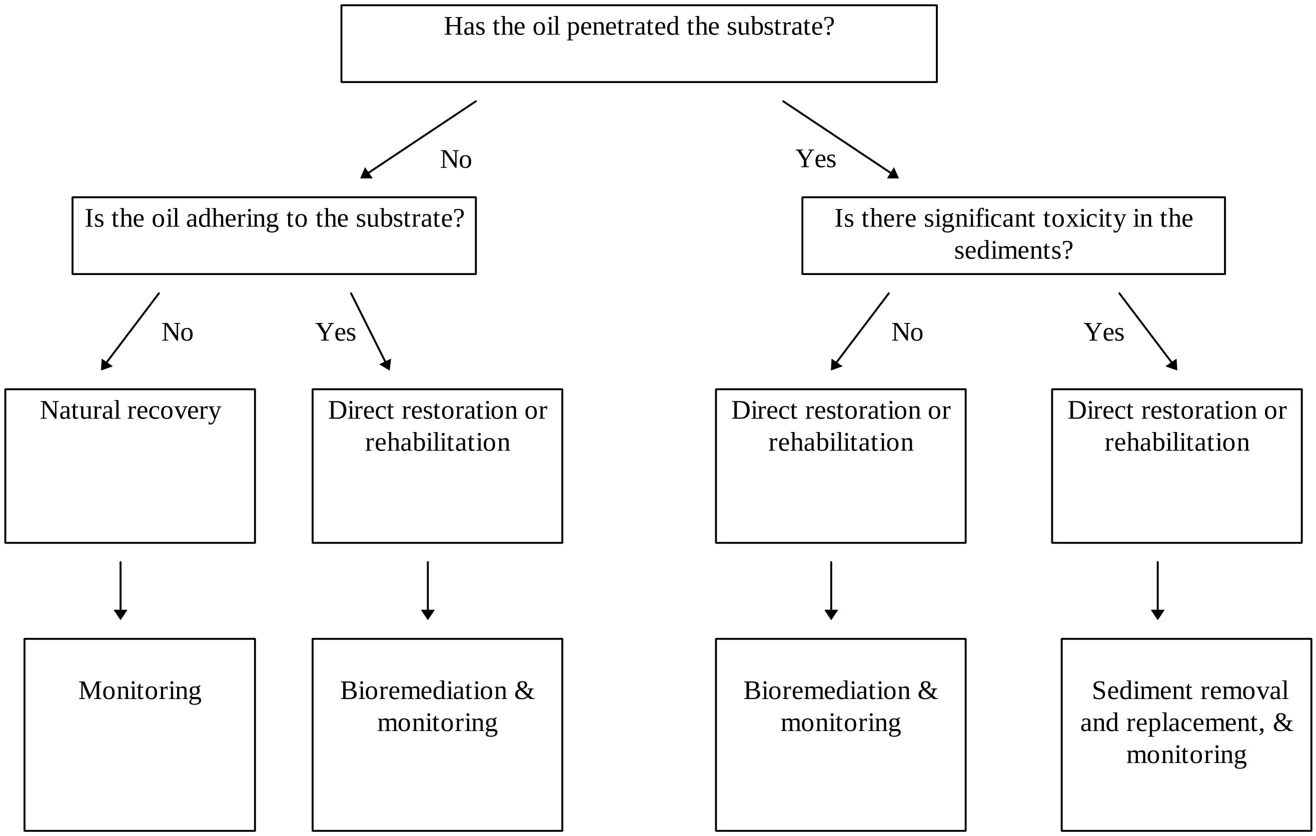

Exhibit 5.26 Decision diagram for the restoration alternatives and actions for estuarine and marine intertidal mud flats and freshwater silt-mud shores.

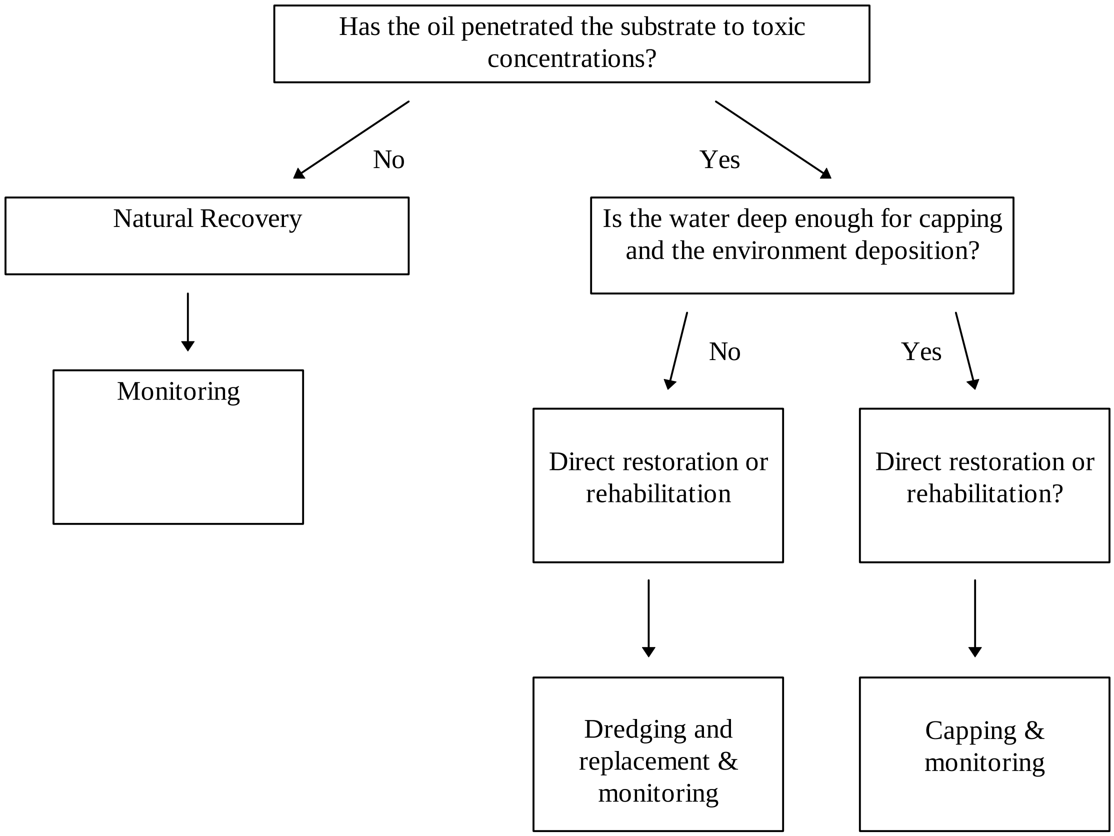

Exhibit 5.27 Decision diagram for the restoration alternatives and actions for estuarine and marine subtidal cobble-gravel, sand and silt-mud bottom, and freshwater sand and silt-mud bottoms.

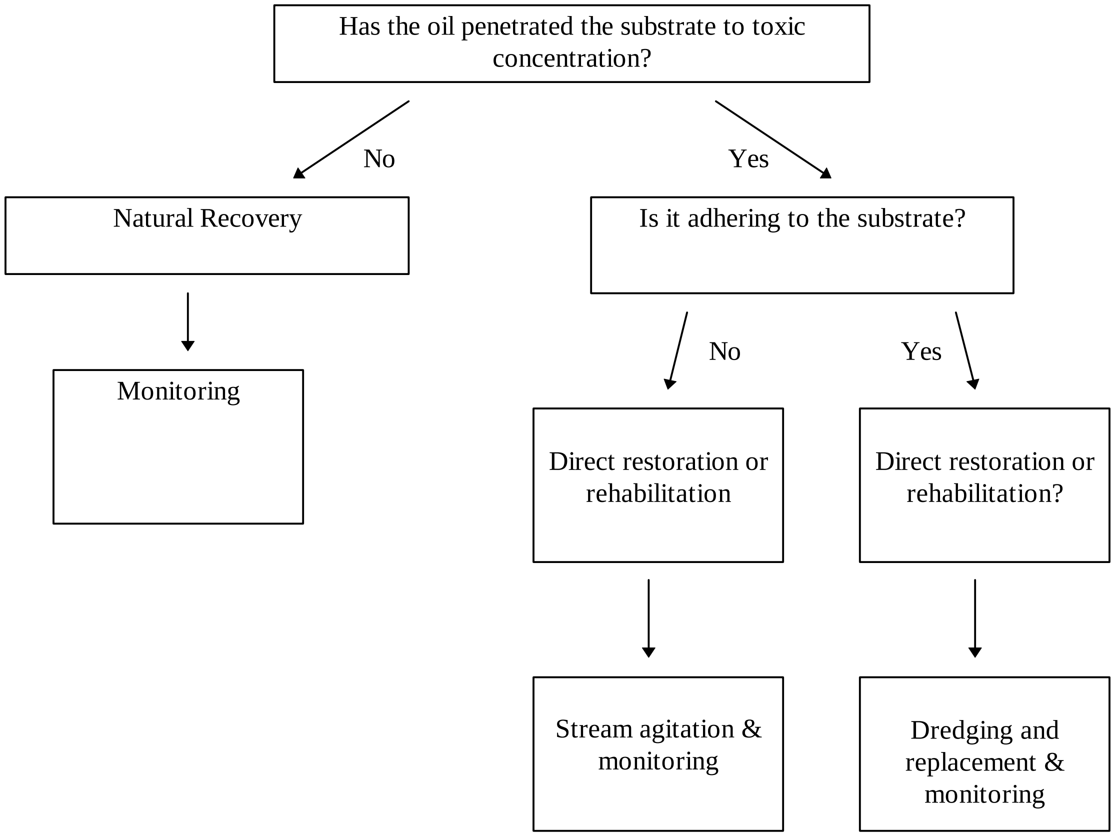

Exhibit 5.28 Decision diagram for the restoration alternatives and actions for freshwater cobble-gravel bottoms.

Exhibit 5.29 Alternatives and actions for restoration of saltmarshes. (M=monitoring costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectivenes s and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	Months to a few years	M
Direct Restoration or Rehabilitation	Replanting, monitoring	None	Yes	Months to a few years	10,000 - 45,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, monitoring	Vegetative cropping from boats	Yes	Months to a few years	32,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, replanting, monitoring	Vegetative cropping from boats	Yes	Months to a few years	42,000 - 77,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, monitoring	Low pressure flushing from boats	Yes	Months to a few years	11,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, replanting, monitoring	Low pressure flushing from boats	Yes	Months to a few years	21,000-56,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing,vegetative cropping, monitoring	Low pressure flushing and vegetative cropping from boats	Yes	Months to a few years	43,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, vegetative cropping, replanting, monitoring	Low pressure flushing and vegetative cropping from boats	Yes	Months to a few years	53,000 - 88,000 + M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Bioremediation from air or boats	Bioremediation in developmental stage	Months to a few years	1300 + M
Direct Restoration or Rehabilitation	Bioremediation, replanting, monitoring	Bioremediation done from air or boats	Bioremediation in developmental stage	Months to a few years	11,000-46,000 + M

Exhibit 5.29 (continued)

Alternative	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, monitoring	Vegetative cropping from boats; Bioremediation from boats or air	Bioremediation in development stage	Months to a few years	33,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, replanting, monitoring	Vegetative cropping from boats; Bioremediation from boats or air	Bioremediation in developmental stage	Months to a few years	43,000-78,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, sediment replacement, replanting, monitoring	Vegetative cropping from boats; Bioremediation from boats or air	Bioremediation in developmental stage; sediment replacement feasible only where equipment has access	Months to a few years	123,000-158,000 + M
Direct Restoration or Rehabilitation	Supplemental erosion control	None	Yes	Months	4-1600 per linear meter + M
Replacement	Enhancement	Appropriate site	Yes	Years, depends on specific actions	highly variable depending on site; monitoring costs should be included
Replacement	Creation	Appropriate site	Yes	Years to decades	highly variable depending on site; monitoring costs should be included

Exhibit 5.30 Alternatives and actions for restoration of mangrove swamps. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	Years to decades	M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Bioremediation from air or boats	Bioremediation in development stage	Decades	1300 + M
Direct Restoration or Rehabilitation	Bioremediation, replanting, monitoring	Bioremediation from air or boats	Bioremediation in developmental stage	Decades	3700-455,000 + M
Direct Restoration or Rehabilitation	Low pressure, flushing, monitoring	Low pressure flushing from boats	Yes	Decades	11,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, replanting monitoring	Low pressure flushing from boats	Yes	Decades	13,000-465,000 + M
Replacement	Enhancement	Appropriate site	Yes	Decades	2,400-454,00 + M
Replacement	Creation	Appropriate site	Yes	Decades	Highly variable; no reported costs; monitoring costs should be included

Exhibit 5.31 Alternatives and actions for restoration of freshwater emergent wetlands. (M=Monitoring costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	Months to years	M
Direct Restoration or Rehabilitation	Replanting, monitoring	None	Yes	Years	11,000-38,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, monitoring	Vegetative cropping from boats	Yes	Years	32,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, replanting, monitoring	Vegetative cropping from boats	Yes	Years	43,000 - 70,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, monitoring	Low pressure flushing from boats	Yes	Years	11,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, replanting, monitoring	Low pressure flushing from boats	Yes	Years	22,000-49,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, vegetative cropping, monitoring	Low pressure flushing and vegetative cropping from boats	Yes	Years	43,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, vegetative cropping, replanting, monitoring	Low pressure flushing and vegetative cropping from boats	Yes	Years	54,000 - 81,000 + M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Bioremediation from air or boats	Bioremediation in developmental stage	Years	1300 + M
Direct Restoration or Rehabilitation	Bioremediation, replanting, monitoring	Bioremediation from air or boats	Bioremediation in developmental stage	Years	12,000-39,000 + M

Exhibit 5.31 (continued)

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, monitoring	Vegetative cropping and bioremediation from boats or air	Bioremediation in developmental stage	Years	33,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, replanting, monitoring	Vegetative cropping and bioremediation from boats or air	Bioremediation in developmental stage	Years	44,000-71,000 + M
Direct Restoration or Rehabilitation	Vegetative cropping, bioremediation, sediment replacement, replanting, monitoring	Vegetative cropping and bioremediation from boats or air	Bioremediation in developmental stage; sediment replacement feasible only where equipment has access	Years	124,000-151,000 + M
Direct Restoration or Rehabilitation	Supplemental erosion control	None	Yes	Years	4-1600 per linear meter + M
Replacement	Enhancement	Appropriate site	Yes	Years	Highly variable depending on site; monitoring costs should be included
Replacement	Creation	Appropriate site	Yes	Years	Highly variable depending on site; monitoring costs should be included

Exhibit 5.32 Alternatives and actions for restoration of freshwater scrub-shrub wetlands. (M=monitoring costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	Months to years	M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Bioremediation from boats or air	Bioremediation in developmental stage	Years	1300 + M
Direct Restoration or Rehabilitation	Bioremediation, replanting, monitoring	Bioremediation from boats or air	Bioremediation in developmental stage	Years	No cost data reported for replanting; costs above apply
Direct Restoration or Rehabilitation	Low pressure flushing, monitoring	Low pressure flushing from boats	Yes	Years	11,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, replanting, monitoring	Low pressure flushing from boats	Yes	Years	No cost data reported for replanting; costs above apply
Replacement	Enhancement	Appropriate site	Yes	Years	No cost data reported; monitoring costs should be included
Replacement	Creation	Appropriate site	Yes	Years	No cost data reported; monitoring costs should be included

Exhibit 5.33 Alternatives and actions for restoration of forested wetlands. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	Decades	M
Direct Restoration or rehabilitation	Bioremediation, monitoring	Bioremediation from boats or air	Bioremediation in development stage	Decades	1300 + M
Direct Restoration or Rehabilitation	Bioremediation, replanting, monitoring	Bioremediation from boats or air	Bioremediation in development stage	Decades	1300-78,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, monitoring	Low pressure flushing from boats	Yes	Decades	11,000 + M
Direct Restoration or Rehabilitation	Low pressure flushing, replanting, monitoring	Low pressure flushing from boats	Yes	Decades	11,000-88,000 + M
Replacement	Enhancement	Appropriate Site	Yes	Decades	Highly variable depending on site; monitoring costs should be included
Replacement	Creation	Appropriate site	Yes	Decades	Highly variable depending on site; monitoring costs should be included

Exhibit 5.34 Alternatives and actions for restoration of intertidal macroalgal beds. (M=monitoring costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	Up to 1 year for minor injury; 5-10 years for great injury	M
Direct Restoration	Vegetative cropping, monitoring	None	Not demonstrated	5-10 years	No cost data identified, monitoring costs should be included
Direct Restoration or Rehabilitation	Replanting, monitoring	None	Not demonstrated	Untested or unknown	No cost data identified, monitoring costs should be included
Direct Restoration or Rehabilitation	Vegetative cropping, replanting, monitoring	None	Not demonstrated	Untested	No cost data identified, monitoring costs should be included
Replacement	Replanting, monitoring	Appropriate site	Not demonstrated	Untested	No cost data identified, monitoring costs should be included

Exhibit 5.35 Alternatives and actions for restoration of kelp beds. (M=Monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	One to several years depending on level of injury	M
Direct Restoration	Vegetative cropping, monitoring	None	Yes	One to several years depending on level of injury	No cost data identified
Direct Restoration or Rehabilitation	Replanting, monitoring	None	Yes	Kelp: $\pm$2 years (depends on planting density, etc.) animal community: unknown	1500-3100 + M
Direct Restoration or Rehabilitation	Replanting, herbivore control, monitoring	None	Feasibility of herbivore control unknown	Kelp: $\pm$2 years (depends on planting density, etc.) animal community: unknown	1500-3100 + M plus costs of herbivore control
Replacement	Off-site planting, monitoring	Appropriate site	Yes	Kelp: $\pm$2 years (depends on planting density, etc.) animal community: unknown	1500-3100 M
Replacement	Off-site planting, herbivore control, monitoring	Appropriate site	Feasibility of herbivore control unknown	Kelp: $\pm$2 years (depends on planting density, etc.) animal community: unknown	1500-3100 + M plus costs of herbivore control

Exhibit 5.36 Alternatives and actions for restoration of seagrass beds. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	1+ year for vegetation; whole community unknown	M
Direct Restoration or Rehabilitation	Replanting, monitoring	Substrate should not be significantly disturbed	Yes	2+ years depending on species, planting density, level of injury to substrate. Animal recovery will vary with availability of nearby sources for migration.	8,000-200,000 + M
Replacement	Off-site replanting, monitoring	Only in previously vegetated sites	Yes	2+ years depending on species, planting density, appropriateness of site selected. Animal recovery will vary with availability of nearby sources for migration.	8,000-200,000 + M

Exhibit 5.37 Alternatives and actions for restoration of freshwater aquatic beds. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	Up to 1 year	M
Direct Restoration	Vegetative cropping, Monitoring	None	Yes	$\pm$1 year	Costs unknown +M
Direct Restoration or Rehabilitation	Replanting, monitoring	None	Availability of appropriate species	1 to several years	Costs unknown +M
Direct Restoration or Rehabilitation	Vegetative cropping, replanting, monitoring	None	Availability of appropriate species	1 to several years	Costs unknown +M
Replacement	Replanting, monitoring	Appropriate site	Availability of appropriate species	1 to several years	Costs unknown +M

Exhibit 5.38 Alternatives and actions for restoration of oyster reefs. (M=Monitoring costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Natural reseeding of oyster bed and monitoring	Natural source of larvae	Yes	1-2 1/2 years	M
Natural Recovery	Flushing, monitoring	Clean water	Yes	Days to weeks?	M
Direct Restoration or Rehabilitation	Reseeding and monitoring	None	Yes	1-2 1/2 years	1200 + M
Direct Restoration or Rehabilitation	Reconstruction, reseeding, monitoring	None	Yes	1-2 1/2 years	3000-15,000 + M
Replacement	Reseeding unproductive area and monitoring	Suitable substrate	Yes	1-2 1/2 years	1200 + M
Replacement	Reconstruction, reseeding, and monitoring	Previous site	Yes	1-2 1/2 years	3000-15000 + M
Replacement	Creation	Appropriate site	Yes	1-2 1/2 years	3000-15000+ M

Exhibit 5.39 Alternatives and actions for restoration of coral reefs. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	1-10 years (longer if damage is extensive)	M
Direct Restoration or Rehabilitation	Coral reconstruction and/or transplants & monitoring	None	Yes	10 years to several decades	2,368,000 + M
Replacement	Off-site coral reconstruction and/or transplants & monitoring	Existing reef with nearby donor site	Yes	10 years to several decades	2,368,000 + M

Exhibit 5.40 Alternatives and actions for restoration of estuarine, marine, and freshwater rocky shores. (M=monitoring, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	Dependent on wave action and oil type. High energy shore-weeks to 5 years; sheltered low energy shore 5-10 years	M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Fertilizer will only remain on shore in low energy areas	Access to shore	To date, no gain in recovery time over natural recovery was demonstrated	24,000-144,000 + M
Direct restoration or rehabilitation	Ambient temperature, low pressure flushing; monitoring	Minimize trampling of biota	Access to shore; availability of equipment	Removes oil without killing additional flora and fauna. Recovery 5-10 years.	52,000-65,000 + M
Direct restoration or rehabilitation	Flushing with chemical agent, monitoring	Minimize trampling of biota	Access to shore; availability of equipment	If non-lethal to biota, recovery in 5-10 years likely.	52,000-65,000 + M
Direct Restoration or Rehabilitation	Hot water, high pressure washing, monitoring	None	Access to shore; availability of equipment	Removes oil but causes further injury to flora and fauna. Longer recover time than for natural recovery.	52,000-65,000 + M
Direct restoration or rehabilitation	Sand blasting, monitoring	None	Access to shore; availability of equipment	Removes oil but causes further injury to flora and fauna. Longer recovery time than for natural recovery.	235,000 + M

Exhibit 5.41 Alternatives and actions for restoration of estuarine, marine, and freshwater cobble-gravel beaches. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	5-10 years	M
Direct Restoration or Rehabilitation	Medium pressure flushing, monitoring	None	Access to beach	Can force oil deeper into substrate and increase recovery time	52,000-65,000 +M
Direct Restoration or Rehabilitation	Sediment washing, monitoring	None	Access to beach, availability of "rock washer"	Causes mortality; recovery rates not yet available	23,000-396,000 + M
Direct Restoration or Rehabilitation	Sediment agitation, monitoring	None	Access to beach	Moves oiled substrate to area of wave action where natural recovery is enhanced.	95,000 + M
Direct restoration or rehabilitation	Bioremediation, monitoring	None	Access to beach	5-10 times faster than natural (1 case in Alaska); still under research	24,000-144,000 + M

Exhibit 5.42 Alternatives and actions for restoration of estuarine, marine, and freshwater sand beaches. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	3-5 years	M
Direct Restoration or Rehabilitation	Flushing	None	Access to beach	Effective in removing oil	52,000-65,000 + M
Direct Restoration or Rehabilitation	Sediment agitation	None	Access to beach	Effective in exposing oiled substrate for natural recovery; 3-5 years after completion	95,000 + M
Direct Restoration or Rehabilitation	Sediment washing	None	Access to beach; availability of sediment washing equipment	Effective in removing oil; no recovery data available.	23,000-247,000 + M
Direct Restoration or Rehabilitation	Substrate removal and replacement	None	Access to beach	Effective in removing oil	106,000 + M
Direct Restoration or Rehabilitation	Bioremediation	None	Access to beach; bioremediation development	Recovery may be better than for natural recovery; time not determined; under research	24,000-144,000 + M
Direct Restoration or Rehabilitation	Incineration, monitoring	None	Access to beach; availability of equipment	3-5 years after completion	860,000-1,110,000 + M

Exhibit 5.43 Alternatives and actions for restoration of estuarine and marine intertidal mud flat and freshwater silt-mud shores. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	1 to 5 years	M
Direct Restoration or rehabilitation	Sediment removal and replacement, monitoring	None	Yes	2 years following restoration action	106,000 + M
Direct Restoration or Rehabilitation	Bioremediation, monitoring	Minimize traffic on substrate	Developmental technique	Likely to be 2-5 years following restoration action	24,000-144,000 + M

Exhibit 5.44 Alternatives and actions for restoration of estuarine, marine, and freshwater rock bottom. (M=monitoring Costs, see text for explanation).

Alternatives	Actions	Restrictions to be Effective	Technical and Operational Feasibility	Effectiveness and Success: Recovery Time	Cost 1992 $/ha
Natural Recovery	Monitoring	None	Yes	1-3 years	M

Exhibit 5.45 Alternatives and actions for restoration of estuarine and marine subtidal cobble-gravel, sand and silt-mud bottoms, and freshwater sand and silt-mud bottoms. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	2-3 years	M
Direct Restoration or rehabilitation	Dredging and replacement, monitoring	None	Yes	2-5 years	0.32 - 20.20/m^3 of material removed plus capping costs; plus costs of monitoring
Direct Restoration or Rehabilitation	Capping, monitoring	Depositional environment, deep water	Yes	2-5 years	1.29 - 4.25/m^3 of capping material; plus costs of monitoring

Exhibit 5.46 Alternatives and actions for restoration of freshwater cobble-gravel bottoms. (M=monitoring costs, see text for explanation).

Alternatives	**Actions**	**Restrictions to be Effective**	**Technical and Operational Feasibility**	**Effectiveness and Success: Recovery Time**	**Cost 1992 $/ha**
Natural Recovery	Monitoring	None	Yes	Recovery within 1 year (1 case study)	M
Direct Restoration or Rehabilitation	Streambed agitation, monitoring	None	Yes	Recovery within 1 year (1 case study)	300 + M
Direct Restoration or Rehabilitation	Dredging and replacement, monitoring	None	Yes	Recovery likely to require 2-3 years	0.32-20.20/m^3 of material removed; plus 1.29-4.25/m^3 for replacement sediments; plus monitoring costs

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Subpart A—Introduction

§ 990.10 Purpose.

The goal of the Oil Pollution Act of 1990 (OPA), 33 U.S.C. 2701 *et seq*., is to make the environment and public whole for injuries to natural resources and services resulting from an incident involving a discharge or substantial threat of a discharge of oil (incident). This goal is achieved through the return of the injured natural resources and services to baseline and compensation for interim losses of such natural resources and services from the date of the incident until recovery. The purpose of this part is to promote expeditious and cost-effective restoration of natural resources and services injured as a result of an incident. To fulfill this purpose, this part provides a natural resource damage assessment process for developing a plan for restoration of the injured natural resources and services and pursuing implementation or funding of the plan by responsible parties. This part also provides an administrative process for involving interested parties in the assessment, a range of assessment procedures for identifying and evaluating injuries to natural resources and services, and a means for selecting restoration actions from a reasonable range of alternatives.

§ 990.11 Scope.

The Oil Pollution Act of 1990 (OPA), 33 U.S.C. 2701 *et seq*., provides for the designation of Federal, state, and, if designated by the Governor of the state, local officials to act on behalf of the public as trustees for natural resources and for the designation of Indian tribe and foreign officials to act as trustees for natural resources on behalf of, respectively, the tribe or its members and the foreign government. This part may be used by these officials in conducting natural resource damage assessments when natural resources and/or services are injured as a result of an incident involving an actual or substantial threat of a discharge of oil. This part is not intended to affect the recoverability of natural resource damages when recoveries are sought other than in accordance with this part.

§ 990.12 Overview.

This part describes three phases of a natural resource damage assessment. The Preassessment Phase, during which trustees determine whether to pursue restoration, is described in subpart D of this part. The Restoration Planning Phase, during which trustees evaluate information on potential injuries and use that information to determine the need for, type of, and scale of restoration, is described in subpart E of this part. The Restoration Implementation Phase, during which trustees ensure implementation of restoration, is described in subpart F of this part.

§ 990.13 Rebuttable presumption.

Any determination or assessment of damages to natural resources made by a Federal, State, or Indian trustee in accordance with this part shall have the force and effect of a rebuttable presumption on behalf of the trustee in any administrative or judicial proceeding under OPA.

§ 990.14 Coordination.

(a) *Trustees*. (1) If an incident affects the interests of multiple trustees, the trustees should act jointly under this part to ensure that full restoration is achieved without double recovery of damages. For joint assessments, trustees must designate one or more Lead Administrative Trustee(s) to act as coordinators.

(2) If there is a reasonable basis for dividing the natural resource damage assessment, trustees may act independently under this part, so long as there is no double recovery of damages.

(b) *Response agencies*. Trustees must coordinate their activities conducted concurrently with response operations with response agencies consistent with the NCP and any pre-incident plans developed under § 990.15(a) of this part. Trustees may develop pre-incident memoranda of understanding to coordinate their activities with response agencies.

(c) *Responsible parties*. (1) *Invitation*. Trustees must invite the responsible parties to participate in the natural resource damage assessment described in this part. The invitation to participate should be in writing, and a written response by the responsible parties is required to confirm the desire to participate.

(2) *Timing*. The invitation to participate should be extended to known responsible parties as soon as practicable, but not later than the delivery of the "Notice of Intent to Conduct Restoration Planning," under § 990.44 of this part, to the responsible party.

(3) *Agreements*. Trustees and responsible parties should consider entering into binding agreements to facilitate their interactions and resolve any disputes during the assessment. To maximize cost-effectiveness and cooperation, trustees and responsible parties should attempt to develop a set of agreed-upon facts concerning the incident and/or assessment.

(4) *Nature and extent of participation*. If the responsible parties accept the invitation to participate, the scope of that participation must be determined by the trustees, in light of the considerations in paragraph (c)(5) of this section. At a minimum, participation will include notice of trustee determinations required under this part, and notice and opportunity to comment on documents or plans that significantly affect the nature and extent of the assessment. Increased levels of participation by responsible parties may be developed at the mutual agreement of the trustees and the responsible parties. Trustees will objectively consider all written comments provided by the responsible parties, as well as any other recommendations or proposals that the responsible parties submit in writing to the Lead Administrative Trustee. Submissions by the responsible parties will be included in the administrative record. Final authority to make determinations regarding injury and restoration rest solely with the trustees. Trustees may end participation by responsible parties who, during the conduct of the assessment, in the sole judgment of the trustees, cause interference with the trustees' ability to fulfill their responsibilities under OPA and this part.

(5) *Considerations*. In determining the nature and extent of participation by the responsible parties or their representatives, trustees may consider such factors as:

(i) Whether the responsible parties have been identified;

(ii) The willingness of responsible parties to participate in the assessment;

(iii) The willingness of responsible parties to fund assessment activities;

(iv) The willingness and ability of responsible parties to conduct assessment activities in a technically sound and timely manner and to be bound by the results of jointly agreed upon studies;

(v) The degree of cooperation of the responsible parties in the response to the incident; and

(vi) The actions of the responsible parties in prior assessments.

(6) Request for alternative assessment procedures.

(i) The participating responsible parties may request that trustees use assessment procedures other than those selected by the trustees if the responsible parties:

(A) Identify the proposed procedures to be used that meet the requirements of § 990.27 of this part, and provide reasons supporting the technical adequacy and appropriateness of such procedures for the incident and associated injuries;

(B) Advance to the trustees the trustees' reasonable estimate of the cost of using the proposed procedures; and

(C) Agree not to challenge the results of the proposed procedures. The request from the responsible parties may be made at any time, but no later than, fourteen (14) days of being notified of the trustees' proposed assessment procedures for the incident or the injury.

(ii) Trustees may reject the responsible parties' proposed assessment procedures if, in the sole judgment of the trustees, the proposed assessment procedures:

(A) Are not technically feasible;

(B) Are not scientifically or technically sound;

(C) Would inadequately address the natural resources and services of concern;

(D) Could not be completed within a reasonable time frame; or

(E) Do not meet the requirements of § 990.27 of this part.

(7) *Disclosure*. Trustees must document in the administrative record and Restoration Plan the invitation to the responsible parties to participate, and briefly describe the nature and extent of the responsible parties' participation. If the responsible parties' participation is terminated during the assessment, trustees must provide a brief explanation of this decision in the administrative record and Restoration Plan.

(d) *Public*. Trustees must provide opportunities for public involvement after the trustees' decision to develop restoration plans or issuance of any notices to that effect, as provided in § 990.55 of this part. Trustees may also provide opportunities for public involvement at any time prior to this decision if such involvement may enhance trustees' decisionmaking or avoid delays in restoration.

§ 990.15 Considerations to facilitate restoration.

In addition to the procedures provided in subparts D through F of this part, trustees may take other actions to further the goal of expediting restoration of injured natural resources and services, including:

(a) *Pre-incident planning*. Trustees may engage in pre-incident planning activities. Pre-incident plans may identify natural resource damage assessment teams, establish trustee notification systems, identify support services, identify natural resources and services at risk, identify area and regional response agencies and officials, identify available baseline information, establish data management systems, and identify assessment funding issues and options. Potentially responsible parties, as well as all other members of the public interested in and capable of participating in assessments, should be included in pre-incident planning to the fullest extent practicable.

(b) *Regional Restoration Plans*. Where practicable, incident- specific restoration plan development is preferred, however, trustees may develop Regional Restoration Plans. These plans may be used to support a claim under § 990.56 of this part. Regional restoration planning may consist of compiling databases that identify, on a regional or watershed basis, or otherwise as appropriate, existing, planned, or proposed restoration projects that may provide appropriate restoration alternatives for consideration in the context of specific incidents.

Subpart B—Authorities

§ 990.20 Relationship to the CERCLA natural resource damage assessment regulations.

(a) *General*. Regulations for assessing natural resource damages resulting from hazardous substance releases under the Comprehensive Environmental Response, Compensation, and Liability Act of 1980, as amended (CERCLA), 42 U.S.C. 9601 *et seq*., and the Federal Water Pollution Control Act (Clean Water Act), 33 U.S.C. 1321 *et seq*., are codified at 43 CFR part 11. The CERCLA regulations originally applied to natural resource damages resulting from oil discharges as well as hazardous substance releases. This part supersedes 43 CFR part 11 with regard to oil discharges covered by OPA.

(b) *Assessments commenced before February 5, 1996*. If trustees commenced a natural resource damage assessment for an oil discharge under 43 CFR part 11 prior to February 5, 1996 they may complete the assessment in compliance with 43 CFR part 11, or they may elect to use this part, and obtain a rebuttable presumption.

(c) *Oil and hazardous substance mixtures*. For natural resource damages resulting from a discharge or release of a mixture of oil and hazardous substances, trustees must use 43 CFR part 11 in order to obtain a rebuttable presumption.

§ 990.21 Relationship to the NCP.

This part provides procedures by which trustees may determine appropriate restoration of injured natural resources and services, where such injuries are not fully addressed by response actions. Response actions and the coordination with damage assessment activities are conducted pursuant to the National Oil and Hazardous Substances Pollution Contingency Plan (NCP), 40 CFR part 300.

§ 990.22 Prohibition on double recovery.

When taking actions under this part, trustees are subject to the prohibition on double recovery, as provided in 33 U.S.C. 2706(d)(3) of OPA.

§ 990.23 Compliance with NEPA and the CEQ regulations.

(a) *General*. The National Environmental Policy Act (NEPA), 42 U.S.C. 4321 *et seq*. and Council on Environmental Quality (CEQ) regulations implementing NEPA, 40 CFR chapter V, apply to restoration actions by federal trustees, except where a categorical exclusion or other exception to NEPA applies. Thus, when a federal trustee proposes to take restoration actions under this part, it must integrate this part with NEPA, the CEQ regulations, and NEPA regulations promulgated by that federal trustee agency. Where state NEPA-equivalent laws may apply to state trustees, state trustees must consider the extent to which they must integrate this part with their NEPA-equivalent laws. The requirements and process described in this section relate only to NEPA and federal trustees.

(b) *NEPA requirements for federal trustees*. NEPA becomes applicable when federal trustees propose to take restoration actions, which begins with the development of a Draft Restoration Plan under § 990.55 of this part. Depending upon the circumstances of the incident, federal trustees may need to consider early involvement of the public in restoration planning in order to meet their NEPA compliance requirements.

(c) *NEPA process for federal trustees*. Although the steps in the NEPA process may vary among different federal trustees, the process will generally involve the need to develop restoration plans in the form of an Environmental Assessment or Environmental Impact Statement, depending upon the trustee agency's own NEPA regulations.

(1) *Environmental Assessment*. (i) *Purpose*. The purpose of an Environmental Assessment (EA) is to determine whether a proposed restoration action will have a significant (as defined under NEPA and § 1508.27 of the CEQ regulations) impact on the quality of the human environment, in which case an Environmental Impact Statement (EIS) evaluating the impact is required. In the alternative, where the impact will not be significant, federal trustees must issue a Finding of No Significant Impact (FONSI) as part of the restoration plans developed under this part. If significant impacts to the human environment are anticipated, the determination to proceed with an EIS may be made as a result, or in lieu, of the development of the EA.

(ii) *General steps*. (A) If the trustees decide to pursue an EA, the trustees may issue a Notice of Intent to Prepare a Draft Restoration Plan/EA, or proceed directly to developing a Draft Restoration Plan/EA.

(B) The Draft Restoration Plan/EA must be made available for public review before concluding a FONSI or proceeding with an EIS.

(C) If a FONSI is concluded, the restoration planning process should be no different than under § 990.55 of this part, except that the Draft Restoration Plan/EA will include the FONSI analysis.

(D) The time period for public review on the Draft Restoration Plan/EA must be consistent with the federal trustee agency's NEPA requirements, but should generally be no less than thirty (30) calendar days.

(E) The Final Restoration Plan/EA must consider all public comments on the Draft Restoration Plan/EA and FONSI.

(F) The means by which a federal trustee requests, considers, and responds to public comments on the Draft Restoration Plan/EA and FONSI must also be consistent with the federal agency's NEPA requirements.

(2) *Environmental Impact Statement*. (i) *Purpose*. The purpose of an Environmental Impact Statement (EIS) is to involve the public and facilitate the decisionmaking process in the federal trustees' analysis of alternative approaches to restoring injured natural resources and services, where the impacts of such restoration are expected to have significant impacts on the quality of the human environment.

(ii) *General steps*. (A) If trustees determine that restoration actions are likely to have a significant (as defined under NEPA and § 1508.27 of the CEQ regulations) impact on the environment, they must issue a Notice of Intent to Prepare a Draft Restoration Plan/EIS. The notice must be published in the Federal Register.

(B) The notice must be followed by formal public involvement in the development of the Draft Restoration Plan/EIS.

(C) The Draft Restoration Plan/EIS must be made available for public review for a minimum of forty-five (45) calendar days. The Draft Restoration Plan/EIS, or a notice of its availability, must be published in the Federal Register.

(D) The Final Restoration Plan/EIS must consider all public comments on the Draft Restoration Plan/EIS, and incorporate any changes made to the Draft Restoration Plan/EIS in response to public comments.

(E) The Final Restoration Plan/EIS must be made publicly available for a minimum of thirty (30) calendar days before a decision is made on the federal trustees' proposed restoration actions (Record of Decision). The Final Restoration Plan/EIS, or a notice of its availability, must be published in the Federal Register.

(F) The means by which a federal trustee agency requests, considers, and responds to public comments on the Final Restoration Plan/EIS must also be consistent with the federal agency's NEPA requirements.

(G) After appropriate public review on the Final Restoration Plan/EIS is completed, a Record of Decision (ROD) is issued. The ROD summarizes the trustees' decisionmaking process after consideration of any public comments relative to the proposed restoration actions, identifies all restoration alternatives (including the preferred alternative(s)), and their environmental consequences, and states whether all practicable means to avoid or minimize environmental harm were adopted (e.g., monitoring and corrective actions). The ROD may be incorporated with other decision documents prepared by the trustees. The means by which the ROD is made publicly

(d) *Relationship to Regional Restoration Plans or an existing restoration project*. If a available must be consistent with the federal trustee agency's NEPA requirements. (Regional Restoration Plan or existing restoration project is proposed for use, federal trustees may be able to tier their NEPA analysis to an existing EIS, as described in §§ 1502.20 and 1508.28 of the CEQ regulations.

§ 990.24 Compliance with other applicable laws and regulations.

(a) *Worker health and safety*. When taking actions under this part, trustees must comply with applicable worker health and safety considerations specified in the NCP for response actions.

(b) *Natural Resources protection*. When acting under this part, trustees must ensure compliance with any applicable consultation, permitting, or review requirements, including but not limited to: the Endangered Species Act of 1973, 16 U.S.C. 1531 *et seq*.; the Coastal Zone Management Act of 1972, 16 U.S.C. 1451 *et seq*.; the Migratory Bird Treaty Act, 16 U.S.C. 703 *et seq*.; the National Marine Sanctuaries Act, 16 U.S.C. 1431 *et seq*.; the National Historic Preservation Act, 12 U.S.C. 470 *et seq*.; the Marine Mammal Protection Act, 16 U.S.C. 1361 *et seq*.; and the Archaeological Resources Protection Act, 16 U.S.C. 470 *et seq*.

§ 990.25 Settlement.

Trustees may settle claims for natural resource damages under this part at any time, provided that the settlement is adequate in the judgment of the trustees to satisfy the goal of OPA and is fair, reasonable, and in the public interest, with particular consideration of the adequacy of the settlement to restore, replace, rehabilitate, or acquire the equivalent of the injured natural resources and services. Sums recovered in settlement of such claims, other than reimbursement of trustee costs, may only be expended in accordance with a restoration plan, which may be set forth in whole or in part in a consent decree or other settlement agreement, which is made available for public review.

§ 990.26 Emergency restoration.

(a) Trustees may take emergency restoration action before completing the process established under this part, provided that:

(1) The action is needed to minimize continuing or prevent additional injury;

(2) The action is feasible and likely to minimize continuing or prevent additional injury; and

(3) The costs of the action are not unreasonable.

(b) If response actions are still underway, trustees, through their Regional Response Team member or designee, must coordinate with the On-Scene Coordinator (OSC) before taking any emergency restoration actions. Any emergency restoration actions proposed by trustees should not interfere with on-going response actions. Trustees must explain to response agencies through the OSC prior to implementation of emergency restoration actions their reasons for believing that proposed emergency restoration actions will not interfere with on-going response actions.

(c) Trustees must provide notice to identified responsible parties of any emergency restoration actions and, to the extent time permits, invite their participation in the conduct of those actions as provided in § 990.14(c) of this part.

(d) Trustees must provide notice to the public, to the extent practicable, of these planned emergency restoration actions. Trustees must also provide public notice of the justification for, nature and extent of, and results of emergency restoration actions within a reasonable time frame after completion of such actions. The means by which this notice is provided is left to the discretion of the trustee.

§ 990.27 Use of assessment procedures.

(a) *Standards for assessment procedures*. Any procedures used pursuant to this part must comply with all of the following standards if they are to be in accordance with this part:

(1) The procedure must be capable of providing assessment information of use in determining the type and scale of restoration appropriate for a particular injury;

(2) The additional cost of a more complex procedure must be reasonably related to the expected increase in the quantity and/or quality of relevant information provided by the more complex procedure; and

(3) The procedure must be reliable and valid for the particular incident.

(b) *Assessment procedures available*. (1) The range of assessment procedures available to trustees includes, but is not limited to:

(i) Procedures conducted in the field;

(ii) Procedures conducted in the laboratory;

(iii) Model-based procedures, including type A procedures identified in 43 CFR part 11, subpart D, and compensation formulas/schedules; and

(iv) Literature-based procedures.

(2) Trustees may use the assessment procedures in paragraph (b)(1) of this section alone, or in any combination, provided that the standards in paragraph (a) of this section are met, and there is no double recovery.

(c) *Selecting assessment procedures*. (1) When selecting assessment procedures, trustees must consider, at a minimum:

(i) The range of procedures available under paragraph (b) of this section;

(ii) The time and cost necessary to implement the procedures;

(iii) The potential nature, degree, and spatial and temporal extent of the injury;

(iv) The potential restoration actions for the injury; and

(v) The relevance and adequacy of information generated by the procedures to meet information requirements of restoration planning.

(2) If a range of assessment procedures providing the same type and quality of information is available, the most cost-effective procedure must be used.

Subpart C—Definitions

§ 990.30 Definitions.

For the purpose of this rule, the term:

Baseline means the condition of the natural resources and services that would have existed had the incident not occurred. Baseline data may be estimated using historical data, reference data, control data, or data on incremental changes (e.g., number of dead animals), alone or in combination, as appropriate.

Cost-effective means the least costly activity among two or more activities that provide the same or a comparable level of benefits, in the judgment of the trustees.

CEQ regulations means the Council on Environmental Quality regulations implementing NEPA, 40 CFR chapter V.

Damages means damages specified in section 1002(b) of OPA (33 U.S.C. 1002(b)), and includes the costs of assessing these damages, as defined in section 1001(5) of OPA (33 U.S.C. 2701(5)).

Discharge means any emission (other than natural seepage), Intentional or unintentional, and includes, but is not limited to, spilling, leaking, pumping, pouring, emitting, emptying, or dumping, as defined in section 1001(7) of OPA (33 U.S.C. 2701(7)).

Exclusive Economic Zone means the zone established by Presidential Proclamation 5030 of March 10, 1983 (3 CFR, 1984 Comp., p. 22), including the ocean waters of the areas referred to as "eastern special areas" in Article 3(1) of the Agreement between the United States of America and the Union of Soviet Socialist Republics on the Maritime Boundary, signed June 1, 1990, as defined in section 1001(8) of OPA (33 U.S.C. 2701(8)).

Exposure means direct or indirect contact with the discharged oil.

Facility means any structure, group of structures, equipment, or device (other than a vessel) which is used for one or more of the following purposes: exploring for, drilling for, producing, storing, handling, transferring, processing, or transporting oil. This term includes any motor vehicle, rolling stock, or pipeline used for one or more of these purposes, as defined in section 1001(9) of OPA (33 U.S.C. 2701(9)).

Fund means the *Oil Spill Liability Trust Fund*, established by section 9509 of the Internal Revenue Code of 1986 (26 U.S.C. 9509), as defined in section 1001(11) of OPA (33 U.S.C. 2701(11)).

Incident means any occurrence or series of occurrences having the same origin, involving one or more vessels, facilities, or any combination thereof, resulting in the discharge or substantial threat of discharge of oil into or upon navigable waters or adjoining shorelines or the Exclusive Economic Zone, as defined in section 1001(14) of OPA (33 U.S.C. 2701(14)).

Indian tribe (or *tribal*) means any Indian tribe, band, nation, or other organized group or community, but not including any Alaska Native regional or village corporation, which is recognized as eligible for the special programs and services provided by the United States to Indians because of their status as Indians and has governmental authority over lands belonging to or controlled by the tribe, as defined in section 1001(15) of OPA (33 U.S.C. 2701(15)).

Injury means an observable or measurable adverse change in a natural resource or impairment of a natural resource service. Injury may occur directly or indirectly to a natural resource and/or service. Injury incorporates the terms "destruction," "loss," and "loss of use" as provided in OPA.

Lead Administrative Trustee(s) (or *LAT*) means the trustee(s) who is selected by all participating trustees whose natural resources or services are injured by an incident, for the purpose of coordinating natural resource damage assessment activities. The LAT(s) should also facilitate communication between the OSC and other natural resource trustees regarding their activities during the response phase.

NCP means the National Oil and Hazardous Substances Pollution Contingency Plan (National Contingency Plan) codified at 40 CFR part 300, which addresses the identification, investigation, study, and response to incidents, as defined in section 1001(19) of OPA (33 U.S.C. 2701(19)).

Natural resource damage assessment (or *assessment*) means the process of collecting and analyzing information to evaluate the nature and extent of injuries resulting from an incident, and determine the restoration actions needed to bring injured natural resources and services back to baseline and make the environment and public whole for interim losses.

Natural resources means land, fish, wildlife, biota, air, water, ground water, drinking water supplies, and other such resources belonging to, managed by, held in trust by, appertaining to, or otherwise controlled by the United States (including the resources of the Exclusive Economic Zone), any state or local government or Indian tribe, or any foreign government, as defined in section 1001(20) of OPA (33 U.S.C. 2701(20)).

Navigable waters means the waters of the United States, including the territorial sea, as defined in section 1001(21) of OPA (33 U.S.C. 2701(21)).

NEPA means the National Environmental Policy Act, 42 U.S.C. 4321 et seq.

Oil means oil of any kind or in any form, including, but not limited to, petroleum, fuel oil, sludge, oil refuse, and oil mixed with wastes other than dredged spoil. However, the term does not include petroleum, including crude oil or any fraction thereof, that is specifically listed or designated as a hazardous substance under 42 U.S.C. 9601(14)(A) through (F), as defined in section 1001(23) of OPA (33 U.S.C. 2701(23)).

On-Scene Coordinator (or *OSC*) means the official designated by the U.S. Environmental Protection Agency or the U.S. Coast Guard to coordinate and direct response actions under the NCP, or the government official designated by the lead response agency to coordinate and direct response actions under the NCP.

OPA means the Oil Pollution Act of 1990, 33 U.S.C. 2701 *et seq*.

Pathway means any link that connects the incident to a natural resource and/or service, and is associated with an actual discharge of oil.

Person means an individual, corporation, partnership, association, state, municipality, commission, or political subdivision of a state, or any interstate body, as defined in section 1001(27) of OPA (33 U.S.C. 2701(27)).

Public vessel means a vessel owned or bareboat chartered and operated by the United States, or by a state or political subdivision thereof, or by a foreign nation, except when the vessel is engaged in commerce, as defined in section 1001(29) of OPA (33 U.S.C. 2701(29)).

Reasonable assessment costs means, for assessments conducted under this part, assessment costs that are incurred by trustees in accordance with this part. In cases where assessment costs are incurred but trustees do not pursue restoration, trustees may recover their reasonable assessment costs provided that they have determined that assessment actions undertaken were premised on the likelihood of injury and need for restoration. Reasonable assessment costs also include: administrative, legal, and enforcement costs necessary to carry out this part; monitoring and oversight costs; and costs associated with public participation.

Recovery means the return of injured natural resources and services to baseline.

Response (or *remove* or *removal*) means containment and removal of oil or a hazardous substance from water and shorelines or the taking of other actions as may be necessary to minimize or mitigate damage to the public health or welfare, including, but not limited to, fish, shellfish, wildlife, and public and private property, shorelines, and beaches, as defined in section 1001(30) of OPA (33 U.S.C. 2701(30)).

Responsible party means:

(a) *Vessels*. In the case of a vessel, any person owning, operating, or demise chartering the vessel.

(b) *Onshore facilities*. In the case of an onshore facility (other than a pipeline), any person owning or operating the facility, except a federal agency, state, municipality, commission, or political subdivision of a state, or any interstate body, that as the owner transfers possession and right to use the property to another person by lease, assignment, or permit.

(c) *Offshore facilities*. In the case of an offshore facility (other than a pipeline or a deepwater port licensed under the Deepwater Port Act of 1974 (33 U.S.C. 1501 *et seq*.)), the lessee or permittee of the area in which the facility is located or the holder of a right of use and easement granted under applicable state law or the Outer Continental Shelf Lands Act (43 U.S.C. 1301-1356) for the area in which the facility is located (if the holder is a different person than the lessee or permittee), except a federal agency, state, municipality, commission, or political subdivision of a state, or any interstate body, that as owner transfers possession and right to use the property to another person by lease, assignment, or permit.

(d) *Deepwater ports*. In the case of a deepwater port licensed under the Deepwater Port Act of 1974 (33 U.S.C. 1501-1524), the licensee.

(e) *Pipelines*. In the case of a pipeline, any person owning or operating the pipeline.

(f) *Abandonment.* In the case of an abandoned vessel, onshore facility, deepwater port, pipeline, or offshore facility, the persons who would have been responsible parties immediately prior to the abandonment of the vessel or facility, as defined in section 1001(32) of OPA (33 U.S.C. 2701(32)).

Restoration means any action (or alternative), or combination of actions (or alternatives), to restore, rehabilitate, replace, or acquire the equivalent of injured natural resources and services. Restoration includes:

(a) *Primary restoration*, which is any action, including natural recovery, that returns injured natural resources and services to baseline; and

(b) *Compensatory restoration*, which is any action taken to compensate for interim losses of natural resources and services that occur from the date of the incident until recovery.

Services (or *natural resource services*) means the functions performed by a natural resource for the benefit of another natural resource and/or the public.

Trustees (or *natural resource trustees*) means those officials of the federal and state governments, of Indian tribes, and of foreign governments, designated under 33 U.S.C. 2706(b) of OPA.

United States and *State* means the several States of the United States, the District of Columbia, the Commonwealth of Puerto Rico, Guam, American Samoa, the United States Virgin Islands, the Commonwealth of the Northern Marianas, and any other territory or possession of the United States, as defined in section 1001(36) of OPA (33 U.S.C. 2701(36)).

Value means the maximum amount of goods, services, or money an individual is willing to give up to obtain a specific good or service, or the minimum amount of goods, services, or money an individual is willing to accept to forgo a specific good or service. The total value of a natural resource or service includes the value individuals derive from direct use of the natural resource, for example, swimming, boating, hunting, or birdwatching, as well as the value individuals derive from knowing a natural resource will be available for future generations.

Vessel means every description of watercraft or other artificial contrivance used, or capable of being used, as a means of transportation on water, other than a public vessel, as defined in section 1001(37) of OPA (33 U.S.C. 2701(37)).

Subpart D—Preassessment Phase

§ 990.40 Purpose.

The purpose of this subpart is to provide a process by which trustees determine if they have jurisdiction to pursue restoration under OPA and, if so, whether it is appropriate to do so.

§ 990.41 Determination of jurisdiction.

(a) *Determination of jurisdiction*. Upon learning of an incident, trustees must determine whether there is jurisdiction to pursue restoration under OPA. To make this determination, trustees must decide if:

(1) An incident has occurred, as defined in § 990.30 of this part;

(2) The incident is not:

(i) Permitted under a permit issued under federal, state, or local law; or

(ii) From a public vessel; or

(iii) From an onshore facility subject to the Trans-Alaska Pipeline Authority Act, 43 U.S.C. 1651, *et seq*.; and

(3) Natural resources under the trusteeship of the trustee may have been, or may be, injured as a result of the incident.

(b) *Proceeding with preassessment*. If the conditions listed in paragraph (a) of this section are met, trustees may proceed under this part. If one of the conditions is not met, trustees may not take additional action under this part, except action to finalize this determination. Trustees may recover all reasonable assessment costs incurred up to this point provided that conditions in paragraphs (a)(1) and (a)(2) of this section were met and actions were taken with the reasonable belief that natural resources or services under their trusteeship might have been injured as a result of the incident.

§ 990.42 Determination to conduct restoration planning.

(a) *Determination on restoration planning*. If trustees determine that there is jurisdiction to pursue restoration under OPA, trustees must determine whether:

(1) Injuries have resulted, or are likely to result, from the incident;

(2) Response actions have not adequately addressed, or are not expected to address, the injuries resulting from the incident; and

(3) Feasible primary and/or compensatory restoration actions exist to address the potential injuries.

(b) *Proceeding with preassessment*. If the conditions listed in paragraph (a) of this section are met, trustees may proceed under § 990.44 of this part. If one of these conditions is not met, trustees may not take additional action under this part, except action to finalize this determination. However, trustees may recover all reasonable assessment costs incurred up to this point.

§ 990.43 Data collection.

Trustees may conduct data collection and analyses that are reasonably related to Preassessment Phase activities. Data collection and analysis during the Preassessment Phase must be coordinated with response actions such that collection and analysis does not interfere with response actions. Trustees may collect and analyze the following types of data during the Preassessment Phase:

(a) Data reasonably expected to be necessary to make a determination of jurisdiction under § 990.41 of this part, or a determination to conduct restoration planning under § 990.42 of this part;

(b) Ephemeral data; and

(c) Information needed to design or implement anticipated assessment procedures under subpart E of this part.

§ 990.44 Notice of Intent to Conduct Restoration Planning.

(a) *General*. If trustees determine that all the conditions under § 990.42(a) of this part are met and trustees decide to proceed with the natural resource damage assessment, they must prepare a Notice of Intent to Conduct Restoration Planning.

(b) *Contents of the notice*. The Notice of Intent to Conduct Restoration Planning must include a discussion of the trustees' analyses under §§ 990.41 and 990.42 of this part. Depending on information available at this point, the notice may include the trustees' proposed strategy to assess injury and determine the type and scale of restoration. The contents of a notice may vary, but will typically discuss:

(1) The facts of the incident;

(2) Trustee authority to proceed with the assessment;

(3) Natural resources and services that are, or are likely to be, injured as a result of the incident;

(4) Potential restoration actions relevant to the expected injuries; and

(5) If determined at the time, potential assessment procedures to evaluate the injuries and define the appropriate type and scale of restoration for the injured natural resources and services.

(c) *Public availability of the notice*. Trustees must make a copy of the Notice of Intent to Conduct Restoration Planning publicly available. The means by which the notice is made publicly available and whether public comments are solicited on the notice will depend on the nature and extent of the incident and various information requirements, and is left to the discretion of the trustees.

(d) *Delivery of the notice to the responsible parties*. Trustees must send a copy of the notice to the responsible parties, to the extent known, in such a way as will establish the date of receipt, and invite responsible parties' participation in the conduct of restoration planning. Consistent with § 990.14(c) of this part, the determination of the timing, nature, and extent of responsible party participation will be determined by the trustees on an incident-specific basis.

§ 990.45 Administrative record.

(a) If trustees decide to proceed with restoration planning, they must open a publicly available administrative record to document the basis for their decisions pertaining to restoration. The administrative record should be opened concurrently with the publication of the Notice of Intent to Conduct Restoration Planning. Depending on the nature and extent of the incident and assessment, the administrative record should include documents relied upon during the assessment, such as:

(1) Any notice, draft and final restoration plans, and public comments;

(2) Any relevant data, investigation reports, scientific studies, work plans, quality assurance plans, and literature; and

(3) Any agreements, not otherwise privileged, among the participating trustees or with the responsible parties.

(b) Federal trustees should maintain the administrative record in a manner consistent with the Administrative Procedure Act, 5 U.S.C. 551-59, 701-06.

Subpart E—Restoration Planning Phase

§ 990.50 Purpose.

The purpose of this subpart is to provide a process by which trustees evaluate and quantify potential injuries (injury assessment), and use that information to determine the need for and scale of restoration actions (restoration selection).

§ 990.51 Injury assessment—injury determination.

(a) *General*. After issuing a Notice of Intent to Conduct Restoration Planning under § 990.44 of this part, trustees must determine if injuries to natural resources and/or services have resulted from the incident.

(b) *Determining injury*. To make the determination of injury, trustees must evaluate if:

(1) The definition of injury has been met, as defined in § 990.30 of this part; and

(2)(i) An injured natural resource has been exposed to the discharged oil, and a pathway can be established from the discharge to the exposed natural resource; or

(ii) An injury to a natural resource or impairment of a natural resource service has occurred as a result of response actions or a substantial threat of a discharge of oil.

(c) *Identifying injury*. Trustees must determine whether an injury has occurred and, if so, identify the nature of the injury. Potential categories of injury include, but are not limited to, adverse changes in: survival, growth, and reproduction; health, physiology and biological condition; behavior; community composition; ecological processes and functions; physical and chemical habitat quality or structure; and public services.

(d) Establishing exposure and pathway. Except for injuries resulting from response actions or incidents involving a substantial threat of a discharge of oil, trustees must establish whether natural resources were exposed, either directly or indirectly, to the discharged oil from the incident, and estimate the amount or concentration and spatial and temporal extent of the exposure. Trustees must also determine whether there is a pathway linking the incident to the injuries. Pathways may include, but are not limited to, the sequence of events by which the discharged oil was transported from the incident and either came into direct physical contact with a natural resource, or caused an indirect injury.

(e) *Injuries resulting from response actions or incidents involving a substantial threat of a discharge*. For injuries resulting from response actions or incidents involving a substantial threat of a discharge of oil, trustees must determine whether an injury or an impairment of a natural resource service has occurred as a result of the incident.

(f) *Selection of injuries to include in the assessment*. When selecting potential injuries to assess, trustees should consider factors such as:

(1) The natural resources and services of concern;

(2) The procedures available to evaluate and quantify injury and associated time and cost requirements;

(3) The evidence indicating exposure;

(4) The pathway from the incident to the natural resource and/or service of concern;

(5) The adverse change or impairment that constitutes injury;

(6) The evidence indicating injury;

(7) The mechanism by which injury occurred;

(8) The potential degree, and spatial and temporal extent of the injury;

(9) The potential natural recovery period; and

(10) The kinds of primary and/or compensatory restoration actions that are feasible.

§ 990.52 Injury assessment—quantification.

(a) *General*. In addition to determining whether injuries have resulted from the incident, trustees must quantify the degree, and spatial and temporal extent of such injuries relative to baseline.

(b) *Quantification approaches*. Trustees may quantify injuries in terms of:

(1) The degree, and spatial and temporal extent of the injury to a natural resource;

(2) The degree, and spatial and temporal extent of injury to a natural resource, with subsequent translation of that adverse change to a reduction in services provided by the natural resource; or

(3) The amount of services lost as a result of the incident.

(c) *Natural recovery*. To quantify injury, trustees must estimate, quantitatively or qualitatively, the time for natural recovery without restoration, but including any response actions. The analysis of natural recovery may consider such factors as:

(1) The nature, degree, and spatial and temporal extent of injury;

(2) The sensitivity and vulnerability of the injured natural resource and/or service;

(3) The reproductive and recruitment potential;

(4(The resistance and resilience (stability) of the affected environment;

(5) The natural variability; and

(6) The physical/chemical processes of the affected environment.

§ 990.53 Restoration selection—developing restoration alternatives.

(a) *General*. (1) If the information on injury determination and quantification under §§ 990.51 and 990.52 of this part and its relevance to restoration justify restoration, trustees may proceed with the Restoration Planning Phase. Otherwise, trustees may not take additional action under this part. However, trustees may recover all reasonable assessment costs incurred up to this point.

(2) Trustees must consider a reasonable range of restoration alternatives before selecting their preferred alternative(s). Each restoration alternative is comprised of primary and/or compensatory restoration components that address one or more specific injury(ies) associated with the incident. Each alternative must be designed so that, as a package of one or more actions, the alternative would make the environment and public whole. Only those alternatives considered technically feasible and in accordance with applicable laws, regulations, or permits may be considered further under this part.

(b) *Primary restoration*. (1) *General*. For each alternative, trustees must consider primary restoration actions, including a natural recovery alternative.

(2) *Natural recovery*. Trustees must consider a natural recovery alternative in which no human intervention would be taken to directly restore injured natural resources and services to baseline.

(3) *Active primary restoration actions*. Trustees must consider an alternative comprised of actions to directly restore the natural resources and services to baseline on an accelerated time frame. When identifying such active primary restoration actions, trustees may consider actions that:

(i) Remove conditions that would prevent or limit the effectiveness of any restoration action (e.g., residual sources of contamination);

(ii) May be necessary to return the physical, chemical, and/or biological conditions necessary to allow recovery or restoration of the injured natural resources (e.g., replacing substrate or vegetation, or modifying hydrologic conditions); or

(iii) Return key natural resources and services, and would be an effective approach to achieving or accelerating a return to baseline (e.g., replacing essential species, habitats, or public services that would facilitate the replacement of other, dependent natural resource or service components).

(c) *Compensatory restoration*. (1) *General*. For each alternative, trustees must also consider compensatory restoration actions to compensate for the interim loss of natural resources and services pending recovery.

(2) *Compensatory restoration actions*. To the extent practicable, when evaluating compensatory restoration actions, trustees must consider compensatory restoration actions that provide services of the same type and quality, and of comparable value as those injured. If, in the judgment of the trustees, compensatory actions of the same type and quality and comparable value cannot provide a reasonable range of alternatives, trustees should identify actions that provide natural resources and services of comparable type and quality as those provided by the injured natural resources. Where the injured and replacement natural resources and services are not of comparable value, the scaling process will involve valuation of lost and replacement services.

(d) *Scaling restoration actions*. (1) *General*. After trustees have identified the types of restoration actions that will be considered, they must determine the scale of those actions that will make the environment and public whole. For primary restoration actions, scaling generally applies to actions involving replacement and/or acquisition of equivalent of natural resources and/or services.

(2) *Resource-to-resource and service-to-service scaling approaches*. When determining the scale of restoration actions that provide natural resources and/or services of the same type and quality, and of comparable value as those lost, trustees must consider the use of a resource-to-resource or service-to-service scaling approach. Under this approach, trustees determine the scale of restoration actions that will provide natural resources and/or services equal in quantity to those lost.

(3) *Valuation scaling approach*. (i) Where trustees have determined that neither resource-to-resource nor service-to-service scaling is appropriate, trustees may use the valuation scaling approach. Under the valuation scaling approach, trustees determine the amount of natural resources and/or services that must be provided to produce the same value lost to the public. Trustees must explicitly measure the value of injured natural resources and/or services, and then determine the scale of the restoration action necessary to produce natural resources and/or services of equivalent value to the public.

(ii) If, in the judgment of the trustees, valuation of the lost services is practicable, but valuation of the replacement natural resources and/or services cannot be performed within a reasonable time frame or at a reasonable cost, as determined by § 990.27(a)(2) of this part, trustees may estimate the dollar value of the lost services and select the scale of the restoration action that has a cost equivalent to the lost value. The responsible parties may request that trustees value the natural resources and services provided by the restoration action following the process described in § 990.14(c) of this part.

(4) *Discounting and uncertainty*. When scaling a restoration action, trustees must evaluate the uncertainties associated with the projected consequences of the restoration action, and must discount all service quantities and/or values to the date the demand is presented to the responsible parties. Where feasible, trustees should use risk-adjusted measures of losses due to injury and of gains from the restoration action, in conjunction with a riskless discount rate representing the consumer rate of time preference. If the streams of losses and gains cannot be adequately adjusted for risks, then trustees may use a discount rate that incorporates a suitable risk adjustment to the riskless rate.

§ 990.54 Restoration selection—evaluation of alternatives.

(a) *Evaluation standards*. Once trustees have developed a reasonable range of restoration alternatives under § 990.53 of this part, they must evaluate the proposed alternatives based on, at a minimum:

(1) The cost to carry out the alternative;

(2) The 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;

(3) The likelihood of success of each alternative;

(4) The extent to which each alternative will prevent future injury as a result of the incident, and avoid collateral injury as a result of implementing the alternative;

(5) The extent to which each alternative benefits more than one natural resource and/or service; and

(6) The effect of each alternative on public health and safety.

(b) *Preferred restoration alternatives*. Based on an evaluation of the factors under paragraph (a) of this section, trustees must select a preferred restoration alternative(s). If the trustees conclude that two or more alternatives are equally preferable based on these factors, the trustees must select the most cost-effective alternative.

(c) *Pilot projects*. Where additional information is needed to identify and evaluate the feasibility and likelihood of success of restoration alternatives, trustees may implement restoration pilot projects. Pilot projects should only be undertaken when, in the judgment of the trustees, these projects are likely to provide the information, described in paragraph (a) of this section, at a reasonable cost and in a reasonable time frame.

§ 990.55 Restoration selection—developing restoration plans.

(a) *General*. OPA requires that damages be based upon a plan developed with opportunity for public review and comment. To meet this requirement, trustees must, at a minimum, develop a Draft and Final Restoration Plan, with an opportunity for public review of and comment on the draft plan.

(b) *Draft Restoration Plan*. (1) The Draft Restoration Plan should include:

(i) A summary of injury assessment procedures used;

(ii) A description of the nature, degree, and spatial and temporal extent of injuries resulting from the incident;

(iii) The goals and objectives of restoration;

(iv) The range of restoration alternatives considered, and a discussion of how such alternatives were developed under Sec. 990.53 of this part, and evaluated under § 990.54 of this part;

(v) Identification of the trustees' tentative preferred alternative(s);

(vi) A description of past and proposed involvement of the responsible parties in the assessment; and

(vii) A description of monitoring for documenting restoration effectiveness, including performance criteria that will be used to determine the success of restoration or need for interim corrective action.

(2) When developing the Draft Restoration Plan, trustees must establish restoration objectives that are specific to the injuries. These objectives should clearly specify the desired outcome, and the performance criteria by which successful restoration will be judged. Performance criteria may include structural, functional, temporal, and/or other demonstrable factors. Trustees must, at a minimum, determine what criteria will:

(i) Constitute success, such that responsible parties are relieved of responsibility for further restoration actions; or

(ii) Necessitate corrective actions in order to comply with the terms of a restoration plan or settlement agreement.

(3) The monitoring component to the Draft Restoration Plan should address such factors as duration and frequency of monitoring needed to gauge progress and success, level of sampling needed to detect success or the need for corrective action, and whether monitoring of a reference or control site is needed to determine progress and success. Reasonable monitoring and oversight costs cover those activities necessary to gauge the progress, performance, and success of the restoration actions developed under the plan.

(c) *Public review and comment*. The nature of public review and comment on the Draft and Final Restoration Plans will depend on the nature of the incident and any applicable federal trustee NEPA requirements, as described in §§ 990.14(d) and 990.23 of this part.

(d) *Final Restoration Plan*. Trustees must develop a Final Restoration Plan that includes the information specified in paragraph (a) of this section, responses to public comments, if applicable, and an indication of any changes made to the Draft Restoration Plan.

Sec. 990.56 Restoration selection—use of a Regional Restoration Plan or existing restoration project.

(a) *General*. Trustees may consider using a Regional Restoration Plan or existing restoration project where such a plan or project is determined to be the preferred alternative among a range of feasible restoration alternatives for an incident, as determined under § 990.54 of this part. Such plans or projects must be capable of fulfilling OPA's intent for the trustees to restore, rehabilitate, replace, or acquire the equivalent of the injured natural resources and services and compensate for interim losses.

(b) *Existing plans or projects*. (1) *Considerations*. Trustees may select a component of a Regional Restoration Plan or an existing restoration project as the preferred alternative, provided that the plan or project:

(i) Was developed with public review and comment or is subject to public review and comment under this part;

(ii) Will adequately compensate the environment and public for injuries resulting from the incident;

(iii) Addresses, and is currently relevant to, the same or comparable natural resources and services as those identified as having been injured; and

(iv) Allows for reasonable scaling relative to the incident.

(2) *Demand*. (i) If the conditions of paragraph (b)(1) of this section are met, the trustees must invite the responsible parties to implement that component of the Regional Restoration Plan or existing restoration project, or advance to the trustees the trustees' reasonable estimate of the cost of implementing that component of the Regional Restoration Plan or existing restoration project.

(ii) If the conditions of paragraph (b)(1) of this section are met, but the trustees determine that the scale of the existing plan or project is greater than the scale of compensation required by the incident, trustees may only request funding from the responsible parties equivalent to the scale of the restoration determined to be appropriate for the incident of concern. Trustees may pool such partial recoveries until adequate funding is available to successfully implement the existing plan or project.

(3) *Notice of Intent To Use a Regional Restoration Plan or Existing Restoration Project*. If trustees intend to use an appropriate component of a Regional Restoration Plan or existing restoration project, they must prepare a Notice of Intent to Use a Regional Restoration Plan or Existing Restoration Project. Trustees must make a copy of the notice publicly available. The notice must include, at a minimum:

(i) A description of the nature, degree, and spatial and temporal extent of injuries; and

(ii) A description of the relevant component of the Regional Restoration Plan or existing restoration project; and

(iii) An explanation of how the conditions set forth in paragraph (b)(1) of this section are met.

Subpart F—Restoration Implementation Phase

Sec. 990.60 Purpose.

The purpose of this subpart is to provide a process for implementing restoration.

§ 990.61 Administrative record.

(a) *Closing the administrative record for restoration planning*. Within a reasonable time after the trustees have completed restoration planning, as provided in §§ 990.55 and 990.56 of this part, they must close the administrative record. Trustees may not add documents to the administrative record once it is closed, except where such documents:

(1) Are offered by interested parties that did not receive actual or constructive notice of the Draft Restoration Plan and the opportunity to comment on the plan;

(2) Do not duplicate information already contained in the administrative record; and

(3) Raise significant issues regarding the Final Restoration Plan.

(b) *Opening an administrative record for restoration implementation*. Trustees may open an administrative record for implementation of restoration, as provided in Sec. 990.45 of this part. The costs associated with the administrative record are part of the costs of restoration. Ordinarily, the administrative record for implementation of restoration should document, at a minimum, all Restoration Implementation Phase decisions, actions, and expenditures, including any modifications made to the Final Restoration Plan.

§ 990.62 Presenting a demand.

(a) *General*. After closing the administrative record for restoration planning, trustees must present a written demand to the responsible parties. Delivery of the demand should be made in a manner that establishes the date of receipt by the responsible parties.

(b) *When a Final Restoration Plan has been developed*. Except as provided in paragraph (c) of this section and in Sec. 990.14(c) of this part, the demand must invite the responsible parties to either:

(1) Implement the Final Restoration Plan subject to trustee oversight and reimburse the trustees for their assessment and oversight costs; or

(2) Advance to the trustees a specified sum representing trustee assessment costs and all trustee costs associated with implementing the Final Restoration Plan, discounted as provided in § 990.63(a) of this part.

(c) *Regional Restoration Plan or existing restoration project*. When the trustees use a Regional Restoration Plan or an existing restoration project under Sec. 990.56 of this part, the demand will invite the responsible parties to implement a component of a Regional Restoration Plan or existing restoration project, or advance the trustees' estimate of damages based on the scale of the restoration determined to be appropriate for the incident of concern, which may be the entire project or a portion thereof.

(d) *Response to demand*. The responsible parties must respond within ninety (90) calendar days in writing by paying or providing binding assurance they will reimburse trustees' assessment costs and implement the plan or pay assessment costs and the trustees' estimate of the costs of implementation.

(e) *Additional contents of demand*. The demand must also include:

(1) Identification of the incident from which the claim arises;

(2) Identification of the trustee(s) asserting the claim and a statement of the statutory basis for trusteeship;

(3) A brief description of the injuries for which the claim is being brought;

(4) An index to the administrative record;

(5) The Final Restoration Plan or Notice of Intent to Use a Regional Restoration Plan or Existing Restoration Project; and

(6) A request for reimbursement of:

(i) Reasonable assessment costs, as defined in § 990.30 of this part and discounted as provided in Sec. 990.63(b) of this part;

(ii) The cost, if any, of conducting emergency restoration under § 990.26 of this part, discounted as provided in Sec. 990.63(b) of this part; and

(iii) Interest on the amounts recoverable, as provided in section 1005 of OPA (33 U.S.C. 2705), which allows for prejudgment and post-judgment interest to be paid at a commercial paper rate, starting from thirty (30) calendar days from the date a demand is presented until the date the claim is paid.

§ 990.63 Discounting and compounding.

(a) *Estimated future restoration costs*. When determining estimated future costs of implementing a Final Restoration Plan, trustees must discount such future costs back to the date the demand is presented. Trustees may use a discount rate that represents the yield on recoveries available to trustees. The price indices used to project future inflation should reflect the major components of the restoration costs.

(b) *Past assessment and emergency restoration costs*. When calculating the present value of assessment and emergency restoration costs already incurred, trustees must compound the costs forward to the date the demand is presented. To perform the compounding, trustees may use the actual U.S. Treasury borrowing rate on marketable securities of comparable maturity to the period of analysis. For costs incurred by state or tribal trustees, trustees may compound using parallel state or tribal borrowing rates.

(c) Trustees are referred to Appendices B and C of OMB Circular A-94 for information about U.S. Treasury rates of various maturities and guidance in calculation procedures. Copies of Appendix C, which is regularly updated, and of the Circular are available from the OMB Publications Office (202-395-7332).

§ 990.64 Unsatisfied demands.

(a) If the responsible parties do not agree to the demand within ninety (90) calendar days after trustees present the demand, the trustees may either file a judicial action for damages or seek an appropriation from the Oil Spill Liability Trust Fund, as provided in section 1012(a)(2) of OPA (33 U.S.C. 2712(a)(2)).

(b) Judicial actions and claims must be filed within three (3) years after the Final Restoration Plan or Notice of Intent to Use a Regional Restoration Plan or Existing Restoration Project is made publicly available, in accordance with 33 U.S.C. 2717(f)(1)(B) and 2712(h)(2).

§ 990.65 Opening an account for recovered damages.

(a) *General*. Sums recovered by trustees in satisfaction of a natural resource damage claim must be placed in a revolving trust account. Sums recovered for past assessment costs and emergency restoration costs may be used to reimburse the trustees. All other sums must be used to implement the Final Restoration Plan or all or an appropriate component of a Regional Restoration Plan or an existing restoration project.

(b) *Joint trustee recoveries*. (1) *General*. Trustees may establish a joint account for damages recovered pursuant to joint assessment activities, such as an account under the registry of the applicable federal court.

(2) *Management*. Trustees may develop enforceable agreements to govern management of joint accounts, including agreed-upon criteria and procedures, and personnel for authorizing expenditures out of such joint accounts.

(c) *Interest-bearing accounts*. Trustees may place recoveries in interest-bearing revolving trust accounts, as provided by section 1006(f) of OPA (33 U.S.C. 2706(f)). Interest earned on such accounts may only be used for restoration.

(d) *Escrow accounts*. Trustees may establish escrow accounts or other investment accounts.

(e) *Records*. Trustees must maintain appropriate accounting and reporting procedures to document expenditures from accounts established under this section.

(f) *Oil Spill Liability Trust Fund*. Any sums remaining in an account established under this section that are not used either to reimburse trustees for past assessment and emergency restoration costs or to implement restoration must be deposited in the Oil Spill Liability Trust Fund, as provided by section 1006(f) of OPA (33 U.S.C. 2706(f)).

§ 990.66 Additional considerations.

(a) Upon settlement of a claim, trustees should consider the following actions to facilitate implementation of restoration:

(1) Establish a trustee committee and/or memorandum of understanding or other agreement to coordinate among affected trustees, as provided in § 990.14(a)(3) of this part;

(2) Develop more detailed workplans to implement restoration;

(3) Monitor and oversee restoration; and

(4) Evaluate restoration success and the need for corrective action.

(b) The reasonable costs of such actions are included as restoration costs.

RELATED GUIDANCE DOCUMENTS APPENDIX B

In support of the NRDA regulations under OPA and for the purpose of facilitating the NRDA process under OPA, NOAA has produced a number of related guidance documents, in addition to the Primary Restoration Guidance Document, that are relevant to the restoration process. All of these documents are currently available in final form.

NOAA. 1996. Preassessment Phase, Guidance Document for Natural Resource Damage Assessment under the Oil Pollution Act of 1990. National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, Silver Spring, MD.

NOAA. 1996. Injury Assessment, Guidance Document for Natural Resource Damage Assessment under the Oil Pollution Act of 1990. National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, Silver Spring, MD.

NOAA. 1996. Specifications for Use of the NRDAM/CME Version 2.4 to Generate Compensation Formulas, Guidance Document for Natural Resource Damage Assessment under the Oil Pollution Act of 1990. National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, Silver Spring, MD.

NOAA. 1996. Restoration Planning, Guidance Document for Natural Resource Damage Assessment under the Oil Pollution Act of 1990. National Oceanic and Atmospheric Administration, Damage Assessment and Restoration Program, Silver Spring, MD.

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C.1 Case Histories of Restoration Planning

In this appendix, case histories of restoration projects are reviewed to give an indication of the state of practice in restoration planning and actions.

C.2 Oil Discharges

C.2.a High Island Oil Discharge

High Island Oil Discharge, High Island, Texas; 5 September 1991

Reference

- Lindsay (1993), Ciccone (1993)

Discharge History

- A pipeline owned and operated by Amoco Pipeline Company ruptured discharging approximately 10,040 gallons of light crude oil into a drainage ditch, barge slip, and adjacent marsh area.

- Amoco responded to the discharge and completed response activities.

- Marshes affected provide habitat for numerous species of waterfowl, shorebirds, songbirds, and terrestrial reptiles and mammals. Aquatic resources affected include commercially and recreationally important finfish and shellfish species, mollusks, invertebrates, and plankton. A fish kill possibly related to the discharge was observed while monitoring response activities.

Restoration Agreement (Process in Reaching Agreement)

- DOI, NOAA, the Texas Parks and Wildlife Department, Texas Water Commission, and Texas General Land Office entered into an agreement with Amoco Pipeline Company for funds directed towards a restoration program.

- The agreement between the trustees and Amoco was called an Administrative Settlement. Amoco agreed to the restoration plan requested by the trustees. By agreeing to this plan, a long-term damage assessment with possible subsequent litigation was averted. This type of agreement between the trustees and discharger has been used at CERCLA sites in the past (i.e., a "restoration" agreement as part of the settlement in lieu of the damage assessment mechanism to achieve restoration).

Restoration Project Description

- The project was to replace open culverts at the Jackson Ditch Road crossing on the Anahuac National Wildlife Refuge in Chambers County, Texas, about 4 miles to the west of the discharge site. Two 60-inch diameter aluminum culverts fitted with flapgrates and flashboard risers were installed. Work was completed in October 1992. The new culverts work as small dikes.

- The goal of the project is to protect and enhance approximately 10,000 acres of intermediate and brackish marsh on the refuge and adjacent private lands by reducing saltwater intrusion and excessive tidal fluctuation, providing water level control, and preventing entry by oil or other hazardous substances discharged in the Gulf Intracoastal Waterway. The project is to provide significant benefits to waterfowl and other migratory birds, preserve vital nursery areas for marine finfish and shellfish, and contribute to the enhancement of surface water resources.

- Oil did reach Jackson Ditch and the marsh adjacent to it. However, this is not a direct habitat restoration project. Instead, this should be termed enhancement because the entire ecosystem is being improved rather than the oiled area being restored.

Restoration Success

- The project appears to be successful although some additional work is necessary. Quantitative monitoring was not performed.

C.2.b. Texaco Well Oil Discharge

Texaco Well Oil Discharge, Lake Salvador, Louisiana (St. Charles Parish); 4 February 1991

Reference

- Lindsay (1993); Louisiana Department of Wildlife and Fisheries (1993)

Discharge History

- Approximately 2,310 gallons of light crude oil was discharged from Texaco well #118.

- The discharge occurred in the southwestern portion of Lake Salvador and escaped containment. It was transported by wind and surface currents approximately 5 miles to the northwest shoreline of the lake. One mile of shoreline and adjacent nearshore locales were affected.

- Lake Salvador averages 5 feet in depth. Water soluble fractions likely dissolved into the water column and distributed through the water column. Less soluble fractions may have deposited onto bottom sediment and adsorbed onto submerged aquatic vegetation.

- Emergency response actions were undertaken by representatives of Texaco, Inc., under the supervision of the U.S. Coast Guard. Response commenced on 5 February and was completed on 25 February. Response actions did not entirely preclude, nor sufficiently remedy, adverse effects to natural resources. A number and variety of natural resources under state and federal trusteeship were injured.

- Resources potentially affected include submerged aquatic vegetation in nearshore areas as well as a variety of benthic organisms in open water areas, both of which provide food for numerous migratory waterfowl including lesser scaup, gadwall, ringed-neck duck, and coot. The lake also supports an extensive commercial and recreational fishery and provides habitat crucial to certain life stages of estuarine-dependent marine fishes and crustaceans.

- Areas potentially affected include Lake Salvador and the natural resources it sustains as well as the wetland natural resources associated with the state of Louisiana's Salvador Wildlife Management Area and Gheens Foundation Golden Ranch Management Area.

Restoration Agreement (Process in Reaching Agreement)

- A Preassessment Screen and Determination was completed for this site by the trustees (USDOI/USFWS, NOAA, LA Dept. of Environmental Quality, LA Dept. of Wildlife and Fisheries). Biological, lake water, geologic, and air resources were shown to be injured or probably injured. For example, 1,048 dead birds were collected, submerged and emerged aquatic vegetation were adversely affected, and 1 mile of the shoreline was affected.

- Data sufficient to pursue a damage assessment was readily available or likely to be obtained at a reasonable cost.

- Texaco reached agreement with the trustees to complete a restoration project through an Administrative Settlement. Similar to the High Island discharge agreement, Texaco agreed to the restoration plan requested by the trustees. By agreeing to this plan, a long-term damage assessment along with possible subsequent litigation was averted.

Restoration Project Description

- The planned restoration project was not at the original discharge site or any location affected by the oil discharge. Hence, this project is replacement, not direct restoration. Work was completed in the "Netherlands" area adjacent to Lake Cataouatche, approximately 6 miles north of the discharge site. This area is a part of the Salvador Wildlife Management Area.

- The project includes approximately 835 feet of piling-tire breakwater that is an addition to a USACOE project being constructed as mitigation for the USACOE West Bank Hurricane Project. This addition completes the structure for the entire "Netherlands" area. Both projects were constructed in a continuous manner and include a total of approximately 4,330 feet of piling-tire breakwater.

- The "Netherlands" area includes 1,500 acres of marsh, cypress ridges, wooded spoil banks, aquatic habitat, and open water within the Salvador Wildlife Management Area. Marsh subsidence and wave exposure has resulted in much erosion. Under present conditions, loss of the aquatic bed, marsh, and woodlands will occur within 25 years.

- The piling-tire breakwater will provide protection from erosion and contribute to sediment deposition. It was projected that such a structure at the Netherlands/Lake Cataouatche interface would maintain existing conditions for the next 50 years.

- Texaco completed the project in late 1991.

Restoration Success

- The project appears to be successful but it is too early to be sure. It was stated by the Louisiana Department of Wildlife and Fisheries that the project is well regarded. However, it will take some time to determine the project's success.

C.2.c Amoco Cadiz Oil Discharge

Amoco Cadiz Oil Discharge, Coastal Marshes in Brittany, France; March 1978

Reference

- Baca (1993), Seneca (1993)

Discharge History

- Much has been written on the Amoco Cadiz oil discharge. Approximately 65,000,000 gallons of oil was lost, much of it washing up along the Brittany shoreline.

- The Ile Grande salt marsh was greatly affected by oil. Marsh was also removed during cleanup operations.

Restoration Agreement (Process in Reaching Agreement)

- There was no agreement between Amoco and any of the local or federal agencies for response, damage assessment, or restoration.

- This restoration project was a result of an invitation from the joint scientific commission of NOAA and Centre National pour l'Exploration des Oceans. Essentially this was a project funded by the U.S. and France.

Restoration Project Description

- During the cleanup, in some areas the above ground marsh vegetation and associated oil was removed, while in other areas the entire marsh surface was stripped including the root mat to a depth of 30 cm. The intertidal creek banks were almost completely lacking in vegetation cover.

- At Ile Grande, marsh vegetation adjacent to the disturbed sites indicated that prior to the oil discharge, the natural marsh was composed primarily of *Juncus maritimus*, *Puccinellis maritima*, *Triglochin maritima*, *Limonium vulgare*, with lesser amounts of *Spartina maritima*. *Halimione portulacoides* was the dominant species along the creek banks prior to the discharge. There was evidence of marsh removal by response operations in the Kerlavos marsh also, but it appeared that the marsh was much less heavily affected than that at Ile Grande.

- Indigenous vegetation was used to restore/rehabilitate part of the Ile Grande Marsh (west of the bridge) and a nearby estuary at Kerlavos. The work (transplanting of vegetation) began in 1979 and continued in 1980 and 1981.

- Lost plants were taken from nearby healthy natural marshes. Later some nursery plants were used. Transplants include plugs (10 to 15 deep cores from 5 to 7 cm diameter composed of root material with attached substrate) and sprigs (root material only).

- Experimental plantings of *Halimione portulacoides, juncus maritimus, Puccinellis maritima, Spartina maritima,* and *Triglochin maritima* were completed. *Triglochin* was a pioneer species and was eliminated after 1979. Over 61 experimental plantings, including over 11,000 transplants, were established.

- Two types of transplants were attempted, conventional and those employing slow release fertilizer. Transplants were completed over a wide range of substrate and elevation conditions.

Restoration Results

- *Spartina* transplants survived at lower elevations better than those of any other species tested.

- The best growth of transplants of all species tested occurred within + or -0.3 m of the natural marsh elevation. The highest survival and growth rates were obtained with *Halimione* and *Puccinellia* transplants. *Puccinellia* was the most successful transplanted species.

- Transplants of *Puccinellia* with a core of root and substrate material intact (plugs) were superior to those transplants with roots only (sprigs) according to survival and growth data. Aboveground growth of this species spread radially at a rate as high as 10 cm annually. At this rate of spread, complete substrate cover would be complete in approximately 3 years after planting.

- Nitrogen and phosphorus (i.e., fertilizer) were required for good transplant growth on disturbed sites (i.e., areas where the root mat was exposed or removed by response operations). Slow release fertilizer materials produced better growth over a wide range of substrate types than did the conventional, more soluble fertilizer materials. Refertilization at various periods after planting produced a significant increase in cover. It is not clear what the value of fertilizer is at oiled, but otherwise, undisturbed sites.

- Sites in the natural marsh, from which transplants were dug, were replanted and became almost completely revegetated within 1 year.

- Other marsh plants invade the plantings more rapidly than they invade unplanted disturbed sites.

Restoration Success

- The marsh restoration of disturbed sites is considered a success. Some initial failure was due to poor transplant locations (i.e., marsh plants placed in tidal flat environments).

- According to Baca et al. (1987), significant revegetation was noted by various workers at the discharge-affected sites at Ile Grande within four years of the discharge, but complete restoration has taken seven to eight years. The extreme cleanup procedures delayed restoration by 2-3 years. However transplanting of indigenous marsh species was beneficial to recovery by establishing open areas and providing attachment substrates for seeds and propagules. Dr. Ernest Seneca, the principal investigator of the transplant study, stated that the transplants reduced the time of full marsh recovery in half (from approximately 10 years to 5 years).

- It is clear that fertilizers help in revegetating disturbed areas but it is not clear what value it provides in those areas oiled but not heavily disturbed by cleanup activities. Seneca noted that only disturbed sites were fertilized. Baca states that fertilizer will help in all transplants.

C.2.d Refinaria Panama Oil Discharge

Restoration of Mangroves following an oil discharge; The 1986 Refineria Panama Oil Discharge

Reference

- Teas (1993)

Discharge History

- A storage tank at the Texaco Refineria Panama on the Caribbean coast of Panama ruptured releasing approximately 50,000 barrels (2,100,000 gallons) of medium light crude oil. Much of the oil accumulated in mangrove-lined bays near the refinery where it killed approximately 75 hectares (185 acres) of mangroves.

- *Rhizophora mangle* is the dominant species and was severely affected.

Restoration Agreement (Process in Reaching Agreement)

- It was believed that the time for natural regeneration of a mangrove forest would be 20 years.

- The Refineria Panama managers were very interested in restoring the killed mangrove forests as soon as practical, so experiments were carried out on techniques that might allow early successful replanting of mangroves in the oiled soil. Because of this urgency, the first replanting experiments began three months after the discharge.

Restoration Project Description

- The mangrove forest was replanted in two ways, seedlings grown in a nursery from propagules and groups of 20 propagules collected from nearby mangroves.

- Propagules planted immediately following the discharge (3-6 months) did poorly due likely to the adherence of droplets of resuspended oil. In addition, planting propagules deep into oiled soil, so roots that formed would be in a sub-surface low oil concentration zone, was ineffective as a restoration technique. After nine months, the propagules did better but the oiled soil still was not suitable for rapid development.

- The protection of propagules from the oiled soil by planting them with upland nursery soil with fertilizer was effective in enhancing growth. The larger the volume of soil, the more growth occurred. Planting of propagules with upland soil was substantially less expensive than growing and planting out nursery seedlings.

- The most effective protection of seedlings was achieved by planting them in dug holes that were lined with plastic and backfilled with upland soil.

- More than 42,000 nursery plants and 44,000 propagules were planted for mangrove forest restoration.

Restoration Success

- The replanting of mangroves in oiled soil using the methods described above was considered a success. Except for a few control areas that were left undisturbed to regenerate naturally, all of the 75 hectares of killed mangroves were replanted with nursery seedlings or propagules within 32 months after the oil discharge. Survival rates are reported to be high.

C.2.e *Exxon Valdez* Oil Discharge

***Exxon Valdez* Oil Discharge (EVOS), Prince William Sound, Alaska, 24 March 1989**

Reference

- Strand (1993), communications with the EVOS Restoration Working Group (1992-1993)

Discharge History

- 11,000,000 gallons of Prudhoe Bay crude oil were discharged at Bligh Reef.

- Surface oil drifted mainly to the southwest and eventually out of Prince William Sound and along the coast.

- Documentation of injuries is extensive in a number of sources. A summary of the injuries may be found in EVOS Trustees (1992a).

- Response was extensive and involved primarily shoreline cleanup (Houghton et al. 1991a,b).

Restoration Planning

- Following the settlement between the six (federal and state) trustees and Exxon (Corporation and Shipping Company) on 8 October 1991, restoration planning has been guided by the Memorandum of Agreement and Consent Decree (Strand et al., 1993).

- A series of documents have been published by the EVOS trustees and Restoration Planning Work Group (RPWG) documenting the restoration planning process (EVOS-RPWG, 1990a, 1990b; EVOS Trustees, 1990c, 1990d, 1991a, 1991b, 1991c, 1992a, 1992b, 1992c, 1992d). These are supplemented by reports by Versar (1990) and the Nature Conservancy (1991). A public information brochure describing alternatives being considered and announcing public meetings was published. Strand et al. (1993) provide a concise review of the process to date.

- Restoration and associated terms are defined in a manner equivalent to the definitions outlined in OPA, as stated in Chapter 1 of this document.

- The goals of the restoration planning effort are (EVOS-RPWG, 1990b):

    - ♦ Identify technically feasible restoration options;

    - ♦ Incorporate an "ecosystem approach" (i.e., broadly focus on recovery of ecosystems, rather than individual components);

    - ♦ Determine rate of natural recovery and where direct restoration may be appropriate;

    - ♦ Encourage, provide for, and be responsive to public participation and review; and

    - ♦ Identify costs of restoration options.

- Restoration must be linked to "consequential injury," i.e., injuries attributable to the *Exxon Valdez* oil discharge and response.

- To maximize the benefits of restoration expenditures, natural recovery is the preferred alternative if the resources appear to be able to recover at a reasonable rate unassisted.

- A list of restoration options was developed from public symposia, meetings, and workshops. Thirty-five candidate options were identified and set forth in the Restoration Framework document (EVOS Trustees, 1992a). These options fell into six possible alternative categories that include no action, management of human uses, manipulation of resources, habitat protection and acquisition, acquisition of equivalent resources, and combination. It is being considered whether alternatives should be prioritized or considered together without prioritization.

- Criteria being used to evaluate the alternatives and options include:

    - ♦ Effects of response or other actions on recovery;
    - ♦ Potential to improve recovery rate;
    - ♦ Feasibility;
    - ♦ Potential effects on human health and safety;
    - ♦ Relationship of expected costs to benefits;
    - ♦ Cost effectiveness;
    - ♦ Consistency with applicable laws;
    - ♦ Potential for additional injury resulting from the option;
    - ♦ Degree to which the option enhances the resource or service;
    - ♦ Degree to which the option benefits more than one resource or service; and
    - ♦ Importance in implementing the option as soon as possible to prevent further injury.

- Habitat protection and acquisition options have received the most public comment. These have been specifically addressed in the Restoration Framework Supplement (EVOS Trustees, 1992b) and The Nature Conservancy (1991). General considerations include:

    - ♦ An established benefit to natural resources injured resulting from habitat protection and acquisition;
    - ♦ Priority for areas under imminent threat;
    - ♦ Cost effectiveness;
    - ♦ Willing sellers; and
    - ♦ Public management requirements.

- A number of pilot restoration studies and other research projects monitoring recovery are being pursued to assist in the planning process, as outlined in the Work Plans (EVOS-RPWG, 1990b; EVOS Trustees, 1991c, 1992a, 1992c)

C.3 Hazardous Waste Sites

C.3.a Wildcat Landfill

Wildcat Landfill CERCLA Site, Delaware

Reference

- Fritz (1993), Wehner (1993)

Discharge History

- The Wildcat landfill site was a sanitary landfill that accepted municipal and industrial waste between 1962 and 1973. Wastes were disposed directly into 44 acres of marsh bordering the St. Jones River resulting in loss of intertidal emergent wetlands and the creation of approximately 5 acres of freshwater, shallow pond, and fringe wetland. The wetland was contaminated by heavy metals and organics from leachate seeps and shallow contaminated groundwater.

- The area supports large turtle and minnow populations and is heavily used by migratory birds for feeding. The Record of the Incident (RI) indicated measurable toxicity and bioaccumulation of heavy metals in fish tissues and turtles, and the potential for adverse food chain effects to migratory birds.

Restoration Agreement (Process in Reaching Agreement)

- Site remediation is being addressed in long-term remedial phases focusing on source control (capping of leachate seeps) and pond cleanup and replacement (draining and filling of the contaminated pond adjacent to the seeps, mitigation for approximately two acres of wetlands surrounding the contaminated pond that will be lost due to capping, continued groundwater monitoring).

- The second Record of Decision (ROD) included much of the restoration plans discussed below. The Remedial Design workplan originally was inconsistent with the ROD and Consent Decree. However, through negotiations between the trustees and responsible parties, the final restoration/mitigation plan (discussed below) was agreed upon. By completing the filling and revegetation of the contaminated pond, the dischargers did not have to institute pumping and treating of groundwater.

Restoration Project Description

- The contaminated pond to be filled will be partially rehabilitated by planting wetland vegetation.

- To compensate for the loss of the wetland associated with filling the contaminated pond, another pond adjacent to the property will be modified to recreate a wetland of equivalent or better habitat value.

- Approximately 2.7 acres of shallow-ponded wetland habitat will be constructed. In addition, a 50-foot floral transition zone will surround the newly created wetland, islands will be created in the pond, and a deed restriction barring construction within 100 feet of the modified pond will be instituted to ensure permanence of the created wetland habitat.

- The primary goal of the restoration is to provide high quality habitat for migratory birds. Because the wetland area to be filled is a shallow freshwater pond, the original restoration plans called for the creation of additional freshwater wetlands. However, the area selected for the mitigation project is intertidally connected to the St. Jones River. In lieu of modifying the system to eliminate its intertidal connection, an intertidal ponded wetland was considered an acceptable restoration alternative because it would serve as equivalent or better habitat for both waterfowl and estuarine fish. This strategy also would help increase the chances for success of the planned wetland creation. As a result, the new modified pond's connection to the river will be enhanced to increase intertidal exchange.

- Additional restoration-related work include provisions for sedimentation control, a maintenance program including control of *Phragmites* spp., field inspections, and long-term monitoring for a period of at least five years to evaluate the success of the wetland vegetation planting.

Restoration Success

- The restoration/mitigation work is to begin in 1992. Hence, no evaluation on its success is possible at this time.

C.3.b Shore Realty Site

Applied Environmental Services/Shore Realty Site, Glenwood Landing, New York

Reference

- Csulak (1993), Wehner (1993)

Discharge History

- Between the 1960's and 1984 the site was used for bulk storage of petroleum products, storage and distribution of chemical solvents, and a hazardous waste storage facility.

- The Record of Decision requires cleanup to include:

    - Vacuum extraction of contaminated unsaturated soils;

    - Collection of contaminated groundwater and treatment by air-stripping;

    - Reinjection of treated groundwater with an indigenous bacteria capable of degrading contaminants in the groundwater and saturated soils; and

    - Treatment (e.g., catalytic oxidation) of contaminant-laden vapors from the vacuum extraction and air-stripping process.

Restoration Agreement (Process in Reaching Agreement)

- Besides the cleanup agreement discussed above, the dischargers (the Group) shall perform a site restoration project along the western and southern shores of the site.

- The restoration described below was written into the consent decree and is the result of the settlement agreement between the Group and the state and federal and natural resource trustees.

- Planting of salt marsh is planned. The planting will be performed in the first appropriate season of the year after the state and the federal trustees (after consultation with the Group) determined that, based on site inspections and sampling carried out, discharges to the shoreline and mudflats adjacent to the site were sufficiently abated by the remedial program to ensure that they are in satisfactory condition to allow for the success of such planting.

- The settlement for marsh restoration is for the amount of $25,000 if initial planting was completed by a certain time frame; $50,000 if no planting is completed by the Group.

- The dischargers will pay the federal trustees $60,000 for the design and implementation of a post-planting monitoring program to determine the functional success of the wetlands restoration. The federal trustees will consult with the state trustee regarding the monitoring program.

- The dischargers will pay the United States on behalf of the federal trustees the sum of $50,000 for the past injury to, destruction of and loss of natural resources, to be used by the federal trustees in accordance with the requirements of CERCLA Section 107(f). The federal trustees will consult with the state trustee with respect to restoration, replacement, or acquisition efforts in New York state.

- The dischargers will pay the sum of $14,000 to the United States on behalf of the federal trustees for past costs incurred by the federal trustees in connection with the site and for the future costs of oversight and participation with respect to the remedy at the site, and oversight of the post-planting monitoring program at the site.

Restoration Project Description

- The area that will be restored historically had a typical assemblage of regional marsh grasses. Presently, it is intertidal mudflat.

- The Group will be required to use proper planting techniques including raking and grading. They will not be required to alter the elevation of the mud flats by dredging, depositing fill material, or other similar means.

- The Group will prepare the described locations for planting and plant juvenile plugs of species such as *Spartina alterniflora*, *Spartina patens,* and/or *Distichlis spicata*, as appropriate.

- Continued planting after the initial planting will be completed at such times as may be necessary to successfully establish such planting. However, the Group will not be required to perform such continued planting after 5 years from the initial planting or if the cost of continued planting exceeds $25,000, whichever comes first.

- The initial planting and any necessary continued planting will be of sufficient quantity and quality to ensure that the planted areas will be self-maintainable and can support marine life indigenous to Hempstead Harbor and Motts Cove marsh areas.

- The natural resource trustees will participate in the development and implementation of the monitoring program called for under the Record of Decision (ROD). At a minimum, monitoring will include the collection of necessary biological data and may incorporate to the appropriate extent results from ongoing federal and state monitoring programs.

Restoration Success

- Restoration (i.e., salt marsh planting) has not yet started because discharges to the shoreline and mudflats are not sufficiently abated by the remedial program. It may be several years before the marsh planting will begin.

C.3.c Commencement Bay Nearshore Tideflats Superfund Site

Commencement Bay Nearshore Tideflats Superfund Site, St. Paul Waterway Area Remedial Action and Habitat Restoration Project, Tacoma, WA

Reference

- Mebane (1993)

Discharge History

- The Commencement Bay ecosystem has received inorganic and organic contaminants from several commercial facilities along the Bay. Contamination has settled into the sediments of Commencement Bay, but little contamination is found upstream in the Puyallup River. The city of Tacoma is one of the Potentially Responsible Parties (RPs) as it is responsible for some contamination from infilling of the Bay for port facilities. Habitats were diked to make way for farms and areas dredged for shipping traffic.

Restoration Agreement (Process in Reaching Agreement)

- A Record of Decision between the dischargers and the EPA/state of Washington was signed in 1989. This followed a very long and tedious negotiation process between EPA, the trustees, and the responsible parties (RPs).

- Presently, a remediation/restoration project was completed only at the St. Paul Waterway. This first cleanup and restoration is an operable unit of the entire Superfund site. In 1987 the trustees and the RPs (Simpson Tacoma Kraft Company, Champion International Corporation) agreed upon the project (built in 1988) with subsequent yearly monitoring. The project includes remedial action and habitat restoration only in the St. Paul Waterway.

- Project approvals under federal and state consent decrees include a long-term Monitoring, Reporting, and Contingency Plan (Monitoring Plan) to ensure the effectiveness of the remedy and provide an annual report of the monitoring results.

- Negotiations between the trustees and RPs to consider further restoration to compensate for injuries at other operable units are still ongoing.

Restoration Project Description

- The restoration follows the sediment remedial action at the St. Paul Waterway adjacent to the Tacoma Kraft Mill. There is some overlap between remediation and restoration throughout this project.

- The restoration project is designed to provide:

  - ♦ Permanent isolation from the environment of chemical contamination found in marine sediments;

  - ♦ Restoration of intertidal and shallow water habitat; and

  - ♦ Monitoring, before and after project construction, to ensure that the remedial action and restoration conformed to the planned design.

- The contaminated sediments in the 17-acre area were capped with clean sediment. This action (i.e., remediation) was integrated with natural resource restoration to produce new intertidal and shallow water habitat in Commencement Bay, which had lost about 90% of such habitat over the last 100 years. More than 6 acres of new intertidal habitat were reconstructed over the portion of the cap along the shoreline. Clean shallow water habitat was provided over the remaining 11 acres. Clean black sand from the mouth of the Puyallup River was used as a cap and promote a new marine habitat.

- The cap is at least 4 feet thick, and 4 to 8 feet thick over the most contaminated area above the high tide line. Varied topography of clean fill was constructed in two areas to allow pools and ridges for diverse habitat. The expectation is that natural forces will continue to redistribute the clean sediments and shape the area.

- Monitoring of the remediation (i.e., the cap) includes physical monitoring, chemical monitoring, and sampling of gas vent, intertidal seep sediments, surface sediments, and subsurface sediments.

- Monitoring of the restoration project includes sampling of the benthic, epibenthic, and algal communities. The biological standard for success consists of not finding:

    - ♦ An adverse effect for benthic infaunal abundance (i.e., mean abundance is less than 50 percent of the reference area);

    - ♦ Amphipod mortality (i.e., mortality exceeding 25 percent of the reference sample); and

    - ♦ Bivalve or echnioderm larval abnormality (i.e., mean abnormality exceeding 20 percent of the reference sample).

- However, it is not clear if these standards would measure injury from contamination leaking through the cap or from restoration failures.

Restoration Success

- The project is now in the long-term or confirmational monitoring phase.

- The 1991 monitoring results indicate the capping project and new habitat are functioning as planned.

- The new habitat is inhabited by diverse biological communities of benthic and epibenthic organisms as well as algae. Shorebirds use the site for feeding and rearing and tide pools observed at low tide are abundant with invertebrates. Productive shoreline habitat continues to be developed at the site where there was essentially no productive habitat three years ago.