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

Hackensack River Meadowlands Hartz Mountain Site

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.

Response to Oiling

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).

Factors Affecting Success of Saltmarsh Restoration

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.

Time-zero report: 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.

Sampling program: 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.

<u>Cypress Creek, Florida</u>

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

<u>Corkscrew Swamp, Florida</u>

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.

<u>Wisconsin Marshes</u>

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.

<u>Milepost 1.9 Jet Fuel Discharge</u>

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.

<u>Milepost 119 Diesel Discharge</u>

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.

<u>Turfs</u>

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.

<u>Individual Mature Plants</u>

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.

Seagrass Recovery

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.

Faunal Recovery

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.