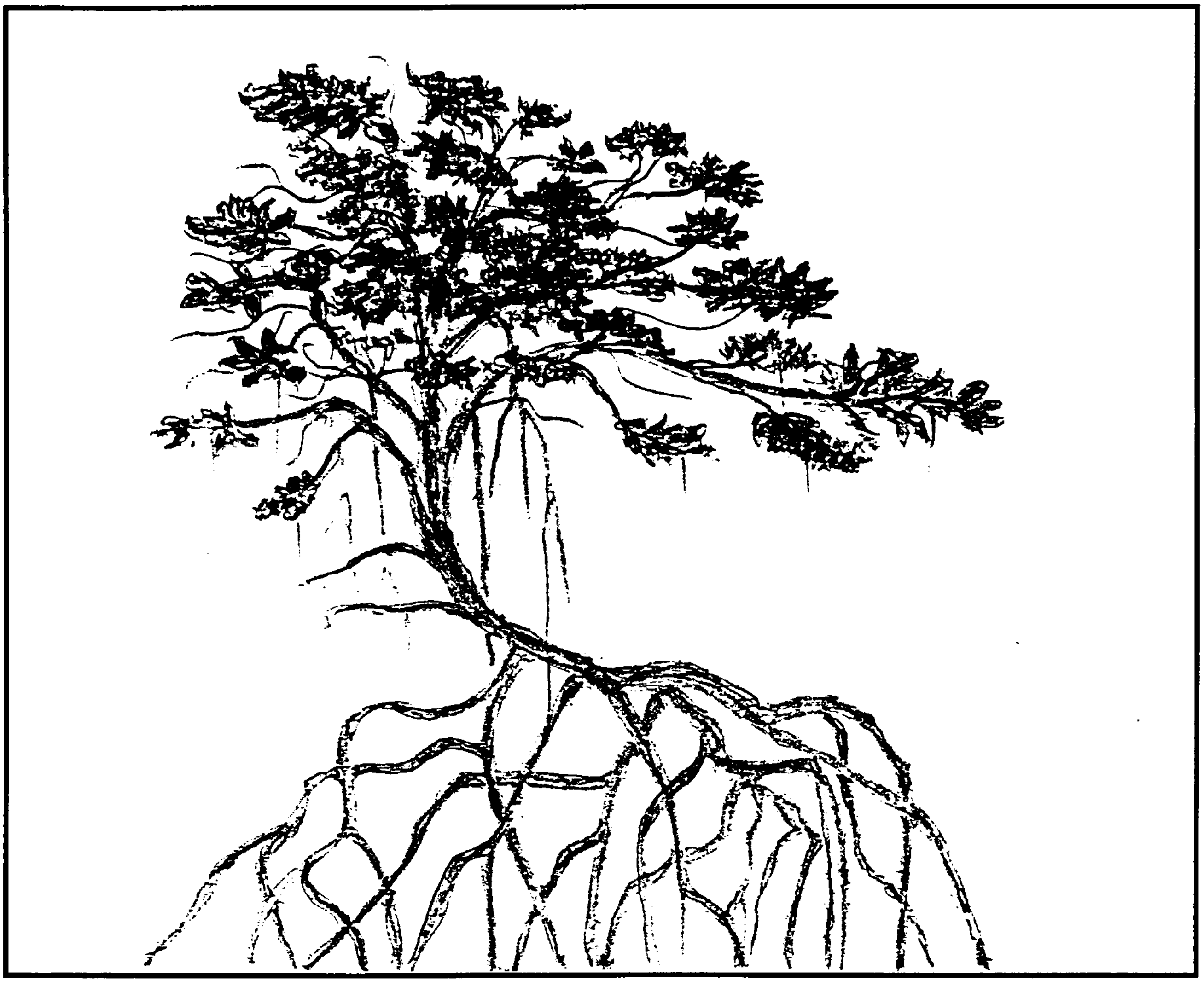
OCS Study
MMS 97-0003

Managing Oil Spills in Mangrove Ecosystems: Effects, Remediation, Restoration, and Modeling



U.S. Department of the Interior
Minerals Management Service
Gulf of Mexico OCS Region

OCS Study
MMS 97-0003

Managing Oil Spills in Mangrove Ecosystems: Effects, Remediation, Restoration, and Modeling

A Review Produced from a Workshop Convened August 1995 at McNeese State University

Editor

C. Edward Proffitt, Ph.D.

Prepared under Cooperative Agreement
14-35-0001-30690
by
Louisiana Environmental Research Center
McNeese State University
Lake Charles, Louisiana 70609

Published by

U.S. Department of the Interior
Minerals Management Service
Gulf of Mexico OCS Region

New Orleans
May 1997

DISCLAIMER

This report was prepared under contract between the Minerals Management Service (MMS) and McNeese State University. This report has been technically reviewed by the editors and by MMS and approved for publication. Approval does not signify that contents necessarily reflect the views and policies of the service, nor does mention of trade names or commercial products constitute endorsement or recommendation for use. It is, however, exempt from review and compliance with MMS editorial standards.

REPORT AVAILABILITY

Copies of this report may be obtained from the Public Information Office at the following address:

U.S. Department of the Interior
Minerals Management Service
Gulf of Mexico OCS Region
Public Information Office (MS 5034)
1201 Elmwood Park Boulevard
New Orleans, Louisiana 70123-2394

Telephone: 1-800-200-GULF
1-504-736-2519

SUGGESTED CITATION

Proffitt, C.E. (ed.). 1996. Managing oil spills in mangrove ecosystems: Effects, Remediation, restoration, and Modeling. OCS Study MMS 97-0003. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. 76 pp.

PREFACE

This document was produced by participants of a Workshop held August 1995 at McNeese State University and sponsored by the U.S. Minerals Management Service. It is a follow-up document to the Proceedings of a Symposium held in New Orleans on July 14-15, 1994 entitled *Gulf of Mexico and Caribbean Oil Spills in Coastal Ecosystems: Assessing Effects, Natural Recovery, and Progress in Remediation Research.* A second workshop held June 1996 addressed coastal marshes and a review paper is being prepared from that meeting as well. The symposium and workshops were supported under a cooperative agreement between MMS and McNeese State University. The sections in this report have been reviewed editorially, but have not been subjected to peer review. Included at the end of the report are outline summaries of discussion sessions held near the end of the symposium.

ACKNOWLEDGMENTS

Mrs. Lucille H. Proffitt drew the artwork for the cover. The McNeese State University Department of Home and Consumer Sciences provided conference rooms and audiovisual equipment for the mangrove workshop.

TABLE OF CONTENTS

EFFECTS OF OIL AND MODELING EFFECTS

CONTAINMENT, CLEANUP, REMEDIATION, AND HABITAT RESTORATION

List of Figures

Page

No.

Oil Spills and Mangroves: An Overview

Samuel C. Snedaker, Patrick D. Biber and Rafael J. Aravjo
Division of Marine Biology and Fisheries
Rosenstiel School of Marine and Atmospheric Science
University of Miami
4600 Rickenbacker Causeway
Miami, Florida 33149-1098 USA

INTRODUCTION

The term mangrove refers to a non-taxonomic grouping of woody, halophytic spermatophytes that occur along low-energy coastlines, deltas, estuaries, and embayments throughout the tropics and subtropics. Tomlinson (1986) recognizes 54 "true or strict" species of which the members of the Rhizophoraceae and Avicenniaceae are the most widely distributed. Since they dominate coastal intertidal areas that are subject to stranding and trapping of oil, a number of researchers (cf. Hayes and Gundlach 1979) consider mangroves to be the most sensitive of all coastal ecosystem types to oil spills. In this regard, Odum and Johannes (1975) speculated that mangroves would take many decades to recover from oil spills. The early research on the topic, however, focused almost exclusively on post-spill damage assessments, in which the primary objectives were to determine the spatial area of impact, and the intensity or degree of acute damage on the impacted flora and fauna (e.g., Lewis 1979, 1980; Getter et al. 1980a,b; Getter et al. 1981). This type of post-spill research has been loosely referred to as the "dripping oil and dead-body count approach." As argued in this review, the observable acute damage following a spill may be insignificant when compared to the longer-term chronic stress induced in mangroves and the contiguous nearshore fauna and flora by the residual oil.

The following summary of acute, secondary and chronic consequences is based primarily on: (1) a review of 28 oil spills in the Caribbean and Gulf of Mexico region (see earlier listing in Getter, Snedaker and Brown 1980c), (2) a number of independent studies (Chan 1977; Page et al. 1979; Gundlach, Scott and Davis 1979; Gundlach et al. 1979; Getter et al. 1980a,b; Odum and Johannes 1975; Hayes, Gundlach and Getter 1980; Lugo, Cintron and Goenaga 1981; Snedaker, Jimenez and Brown 1981; Garrity, Levings and Burns 1994; Burns et al. 1994; Levings, Garrity and Burns 1994), (3) reports of experimental research findings (Jagtap and Untawale 1980; Getter 1983; Ballou et al. 1987; Thomas 1987; Rielinger 1991; Teas et al.,1993), (4) personal observations of the author, and (5) personal communications with knowledgeable persons.

EFFECTS OF STRANDED OIL IN THE MANGROVE ECOSYSTEM

Acute Consequences

The effects of the physical stranding of oil in intertidal mangrove habitats is largely dependent on the oil type, the elapsed time between a spill and its stranding, wind and current conditions, and tidal stage. With regard to oil type, the more highly refined products tend to be relatively more toxic, but because they are also relatively volatile, they are quickly dissipated. The volatile fractions (e.g., naphtha, benzene) are also lost from the heavier oils that remain at sea for extended periods of time prior to stranding. In these regards, the stranding of relatively weathered oils that are depleted in the lighter fractions has a lesser potential to produce acute toxic effects than "fresh" oil and the highly refined products. Whether or not refugee oil eventually becomes stranded along the shoreline is highly dependent on the ambient wind and current conditions. For example, longshore winds and currents tend to move oil parallel to the shoreline, painting long stretches of the seafront, whereas strong onshore winds tend to push the oil onto a smaller length of shoreline but also further inland. Similar to wind-driven oil, oil arriving at the shoreline on incoming tides has the potential to reach deeper into intertidal and supratidal habitats. However, it also has a greater potential for washout on the retreating tide except when trapped in paludal depressions inland from the shoreline fringe. Tidal patterns are particularly important in the context of potential stranding. For example, in south Florida mean sea level is some 20 to 30 cm higher during the Fall than in the Spring (Wanless 1982) meaning that higher water levels in the latter part of the year contribute to a greater potential for inland stranding and trapping, but may also contribute to washout and removal from the intertidal zone (vide Levings et al. 1994).

Mangrove mortality tends to be highest among propagules, seedlings and juvenile trees, due to their proximity to the oil spill surface, and the potential for heavy and repeated oiling on both incoming and outgoing tides. Notwithstanding the potential for relatively intense oiling, shoreline seedlings sometimes survive the initial oiling event. Lugo, Cintron and Goenaga (1981) suggest that mangrove seedlings may be more stress resistant than adult trees based in parts on their field observations and certain of the physiological differences reported by McMillan (1974). In this regard, young-of-the-year seedlings of *Rhizophora* and *Avicennia* utilize cotelydenary reserves prior to developing an extensive root system. This may mitigate against uptake of toxic compounds.

The reported rapid mortality of mangroves following a spill is assumed to be due to mechanical suffocation and the cessation of gas exchange processes associated with the rhizosphere. However, this is somewhat equivocal for two reasons. First, one of us (SCS) has observed and photographed, heavily-oiled prop roots having clean, and presumably active lenticels. To the extent that gas exchange is relatively unaffected, "suffocation" may not be the primary cause of mortality. Secondly, in laboratory experiments using freshly excised *A. germinans* pneumatophores (which also have gas-exchange lenticels), nitrogen gas (N_2) was still able to be transported through the oil film covering pneumatophores (Melvin S. Brown, pers. comm.). Rielinger's (1991) followup work on O_2 uptake and CO_2 fluxes in excised pneumatophores indicated that oil viscosity was the principal factor in gas exchange; the heavier oils reduced exchange to a greater extent than lighter oils.

Further in this regard, since heavy oiling of roots, pneumatophores, tree trunks and sediment can cause direct mortality (or be highly damaging, see 1994 article by Levings, Garrity and Burns), an experiment was conducted on the effects of crude oil (Prudhoe Bay) and a dispersant in Panama in December 1984 (Ballou et al. 1987). In that TROPICS (Tropical Oil Pollution Investigations) field experiment, coastal sites with contiguous mangroves, seagrass, and corals were acutely exposed to untreated crude oil and chemically dispersed crude oil. After 2.5

years of monitoring, the authors concluded that untreated crude oil had severe, long-term effects on the intertidal mangroves and associated fauna, but only minimal effect on the subtidal (continuously submerged) seagrasses and corals. In contrast, the chemically dispersed oil minimized damage to intertidal mangroves but caused "relatively severe, long-term effects on the coral and seagrass environments". In a subsequent study ten years later (Dodge et al., in press), effects of the crude oil on mangroves were still evident in terms of a lower than expected increase in the mean trunk diameter of affected trees, a reduced canopy foliage density and corresponding increased canopy light transmission, increased leaf thickness, and altered patterns of new leaf production and senescent leaf loss.

More recently, Teas et al. (1993) tested one of the newer, less toxic, non-dispersing shoreline "cleaners" (Corexit 9580) on experimentally oiled (bunker C fuel oil) red mangroves (*R. mangle*). They concluded that it was effective in minimizing morbidity and mortality but only if used within a few days of an oiling event. However, in a more recent field study, Quilici et al. (1995) concluded that shoreline cleaners negatively affected the productivity of *R. mangle* leaves on the treated trees, suggesting a sublethal effect not observed in acute mortality studies. Since there are now a number of non-dispersing cleaners on the market, research is needed to evaluate both their efficacy and sublethal chronic effects on other mangrove species, and subtidal organisms, such as seagrass and coral communities.

One acute and severe consequence of the mass mortality of mangroves following a spill event is the death and decomposition of the underground root mass. Since the root mass in the *Rhizophora* and the *Avicennia* represents 40 to 60 percent of the total forest biomass (Snedaker 1995: Snedaker et al. 1994), its loss results in significant subsidence in peat environments and erosion in sandy environments. Nine years following the TROPICS oiling experiment, the forest destruction and the ground surface elevation loss (-8.8 +/- 0.69 cm) were so great that the affected site exhibited the appearance of having been subjected to an "explosion" (Bart Baca, pers. comm.), an observation that was confirmed during the ten-year follow-up study (Dodge et al. in press).

Secondary Consequences

Mangrove mortality and the expression of stress symptoms may be delayed for one to several years period following an oil spill for reasons that may be related to: (1) the persistence of toxic petrogenic compounds in the sediment, and (2) the weakened state of the trees that makes them susceptible to other stressors. For example, following the M/V Howard Star spill in October, 1978, in Tampa, Bay, Florida (Lewis 1979; Getter et al. 1980c), 25 heavily oiled mangrove trees were tagged for long-term monitoring. However, only three of the identified trees exhibited a response (one died and two exhibited stress symptoms), whereas others, not originally identified as heavily oiled, died in mass (Lewis 1980). One of us (SCS) also observed a number of non-tagged dead trees at the same spill site following a severe freeze three years later in January of 1981.

In the earlier M/V Zoe Colocotroni spill in Puerto Rico, observers reported mangrove recruitment at the spill site, and presumed that it was evidence of recovery that later proved to be incorrect. In that instance, mangrove propagules from unaffected or surviving trees colonized the area and developed to the point where their cotyledonous reserves were exhausted; they then perished in mass (Ariel Lugo, pers. comm.). Also, in the subsequent spill litigation, confusion surrounded the question of the exact cause of mortality since the affected area was also subject to natural hypersalinity (Commonwealth of Puerto Rico 1973, 1978; Lugo et al. 1981). These examples suggest that otherwise healthy-appearing mangroves that are exposed to other stressors have a significantly higher probability of morbidity and mortality following exposure to oil (see supporting review and citations in Lugo et al. 1981).

Although the majority of the oil spill literature documents the adverse effects of oil on mangroves, some workers have documented an apparent "stimulating" effect (cf. Page et al. 1979; Thomas 1987). Thomas (1987) recorded an apparent stimulation in 28 experimental treatments versus inhibitory responses in 75 experimental treatments (Appendix A). It was also observed at the TROPICS site in Panama that mangroves regenerating on oil contaminated sediments were healthier and growing faster than similar-sized *R. mangle* at a nearby control station (Dodge et al., in press). Notwithstanding, Thomas argued that any deviation from the "normal" condition is an indication that the oil interfered with normal growth processes and development patterns (see also Getter 1983). To date no one appears to have addressed any of the ecophysiological questions that explicitly pertain to how oil interacts with the physiology of mangroves over the longer term.

Chronic Consequences

As stated earlier, the majority of the research on oil spills focuses on the immediate post-spill acute effects usually because of, and in preparation for, impending litigation. With the notable exception of the published research years after the large oil spill at Bahia las Minas, Panama, in 1986 (cf. Duke and Pinzon 1993; Garrity et al. 1994; Burns et al. 1994; Levings et al. 1994), almost nothing is documented concerning the long-term impact of oil spills in mangrove-dominated habitats. In that context, one has to wonder how legal damage levies (based on acute impact) would be altered if the chronic effects were also taken into account.

In addition to the above citations, the persistence of oil in sediments at oil spill sites years to decades after the spill event has been noted by a number of authors and observers (Mackin 1950; Corredor et al. 1990; Teal et al. 1992; Burns et al. 1994). Disturbance of the oil impregnated sediments, for example, by tidal action, storm pounding, decomposition of the organic substrate, or walking on the sediment (Teas et al. 1989), causes the release of oil in the form of a surface sheen or slick that was termed "bleedwater" by Mackin (1950). Whereas this phenomenon appears to be common in coastal marshes and mangroves, no one appears to have made a determination of the composition of the refugee bleedwater or speculated on the consequences for intertidal and nearshore marine life. Notwithstanding, oil in the form of bleedwater slicks has the ability to scavenge and concentrate organochlorine pesticides and other polar toxic water-insoluble compounds including chellated metals (Seba and Corcoran 1969; Hartung and Klingler 1970; Harvey et al. 1972; Hardy et al. 1987; von Westernhagen et al. 1987). This chemical scavenging and concentrating process could presumably cause the refugee oil bleedwater to become increasingly more toxic over time.

With regard to the heavy metal components in oil, particularly in crude, no one appears to have examined this factor, notwithstanding that mangroves can take up and concentrate metals in leaf tissue at concentrations significantly above background (cf. Carter et al. 1973; Tripp and

Harriss 1975, Peterson et al. 1979; Walsh et al. 1979; Snedaker and Brown 1981, Thomas and Ong 1981, Harbison 1986). Snedaker and Brown (1981), for example, reported the following metal concentration factors (relative to ambient concentrations) for leaves of mature mangroves in southeast Florida: chromium 5-6x, copper 1-2x, iron 2-3x, lead 4-5x, manganese 3-4x, nickel 4-5x, and zinc 1-2x. Although mangroves are relatively resistant to metal toxicity (Walsh et al. 1979), leaves enriched in metals could represent a source of metals entering detrital based marine food webs.

Similar to the metals, the uptake and accumulation of the aromatic fractions in spilled oil might also represent a subtle contamination of the leaf detritus food resource. In an experimental study, Thomas (1987) found that *R. mangle* synthesizes a range of biogenic aliphatic hydrocarbons from C14 to C29 dominated by odd-number compounds with the highest concentrations occurring among C23, C25 and C27-29 compounds. In contrast, the natural presence of aromatic compounds was found to be low. In the experimental dosing study, Thomas found a number of aromatic compounds in leaf tissues and that they closely matched the aromatic composition of the treatment oil. It was also reported that tissue concentrations were inversely proportional to the molecular weight of individual aromatic compounds.

SUMMARY

The principal conclusions that can be drawn from published articles and reports of post-spill evaluations and damage assessments are summarized as follows:

* Light weight refined oils and the volatile components of the heavier oils, including crude oil, are the most toxic, but are also the most susceptible to rapid dissipation, mainly by vaporization.

* Refugee oil that enters and leaves the mangrove habitat on the surface of tidal waters, "paints" the vegetation above the substrate but causes minimal ecological damage.

* Refugee oil that is stranded by wind or current within the mangrove habitat over successive tidal cycles, comes in direct contact with the sediment and surface litter where it becomes trapped. Over time, the oil moves downward into the sediment by gravity and tide-driven hydraulic draw down, where it may persist for decades.

* Oil impregnated sediments continually release small quantities of oil in the form of bleedwater, the ecological consequences of which are unknown.

* The removal of oil impregnated sediments to mitigate against acute and chronic damage is considered to necessarily be more damaging to the habitat than taking no action. Likewise the use of the new shoreline cleaners, once thought to be a viable option, has recently been questioned.

* The most visible ecological effect of stranded oil is the acute mortality of juvenile and adult mangrove trees and the associated fauna, within a period of several days.

* Juvenile and adult mangrove trees that survive an oiling event may experience morbidity and mortality if exposed to an unrelated stressor such as severe cold temperatures or draught.

* Sublethal effects of an oil stranding event are still detectable and measurable in a mangrove forest habitat for years to over a decade later even when the forest visually appears to have fully and completely recovered.

* Results from experiments and field assessments indicate that under certain circumstances, albeit poorly defined, petrogenic compounds can have a significant stimulating effect on mangrove growth and development.

* Because of the long-term ecological consequences, actions, such as booming and skimming, to prevent oil stranding within the mangrove habitat is often the preferred counter measure to be taken in the event of an oil spill.

LITERATURE CITED

Ballou, T.G., R.E. Dodge, S.C. Hess, A.H. Knap and T.D. Sleeter. 1987. Effects of a dispersed and undispersed crude oil on mangroves, seagrasses and corals. Report for American Petroleum Institute. Research Planning Institute, Inc., Columbia, South Carolina, and Bermuda Biological Station for Research, Bermuda.

Burns, K.A., S.D. Garrity, D. Jorissen, J. MacPherson, M. Stoelting, J. Tierney and L. Yelle-Simmons. 1994. The Galeta oil spill. II. Unexpected persistence of oil trapped in mangrove sediments. Estuar. Coast. Shelf Sci. 38:349-364.

Carter, M.R., L.A. Burns, T.R. Cavinder, K.R. Dugger, P.L. Fore, D. B. Hicks, H.L. Revells and T.W. Schmidt. 1973. Ecosystems analysis of Big Cypress Swamp and Estuaries. EPA 904/9-74-002. U.S. Environmental Protection Agency, Region IV, Atlanta. 374 p.

Chan, I.E. 1977. Oil pollution and tropical littoral communities: biological effects of the 1975 Florida Keys oil spill, p. 539-542. In J.O. Ludwigson (ed.) Proceedings of the 1977 Oil Spill Conference. American Petroleum Institute, U.S. Environmental Protection Agency and U.S. Coast Guard. API Publ. No. 4284. 640 p.

Commonwealth of Puerto Rico and the Environmental Quality Board of the Commonwealth of Puerto Rico vs. the SS Zoe Colocotroni, her engines, appurtenances, etc., et al. 1973. Civil No. 252-73. United States District Court for the District of Puerto Rico. San Juan, Puerto Rico.

Commonwealth of Puerto Rico vs. the SS Zoe Colocotroni, Civil No. 252-73. Federal District Court of Puerto Rico, San Juan, Puerto Rico. August 15, 1978.

Corredor, J.E., J.M. Morell and C.E. Castillo. 1990. Persistence of spilled crude oil in a tropical intertidal environment. Mar. Poll. Bull. 21:385-388.

Dodge, R.E., B.J. Baca, A.H. Knap, S.C. Snedaker, and T.D. Sleeter. 1995. The effects of crude oil and dispersed oil in tropical ecosystems: 10 years of monitoring experimental sites. Marine Spill Response Corporation Technical Report Series 95-x, Washington, D.C. [in press]

Duke, N.C. and Z.S. Pinzon. M. 1993. Mangrove forests, pp. 447-533. In D.B. Keller and J.B.C. Jackson (eds.) Long-term assessment of the Oil Spill at Bahia las Mina, Panama, Synthesis Report. Vol. II: Technical Report. U.S. Department of Interior, Minerals Management Service, Gulf of Mexico OCS Regional Office. New Orleans, Louisiana.

Garrity, S.D. S.C. Levings and K.A. Burns. 1994. The Galeta oil spill: I. Long-term effects on the physical structure of the mangrove fringe. Estuar. Coast. Shelf. Sci. 38:327-348.

Getter, C.D. 1983. A laboratory determination of the sensitivity of mangrove seedlings to oil and dispersant combinations. Report submitted to the Exxon Oil Spill Chemicals, Fate and Effects Environmental Subcommittee, RPI/R/82/10/28-30. Research Planning Institute, Inc., Columbia, South Carolina.

Getter, C.D. J. Michel, E.R. Gundlach and G.I. Scott. 1980a. Biological changes of mangrove and sand beach communities at the Peck Slip oil spill site, eastern Puerto Rico. Report. Office Marine Pollution Assessment. NOAA. 63 p.

Getter, C.D., J.M. Nussman, E.R. Gundlach and G.I Scott. 1980b. Biological changes of mangroves and sand beach communities at the Peck Slip oil spill site, eastern Puerto Rico. RPI Report to NOAA. RPI/R/80/2/18-6. Office of Marine Pollution Assessment, Boulder, Colorado. Research Planning Institute, Inc., Columbia, South Carolina. 63 p.

Getter, C.D., S.C. Snedaker and M.S. Brown. 1980c. Assessment of biological damages at the Howard Star oil spill site, Hillsborough Bay and Tampa Bay, Florida. Report. Florida Department of Natural Resources, Tallahassee. 65 p.

Getter, C.D., G. I. Scott and J. Michel. 1981. The effect of oil spills on mangrove forests, a comparison of five oil spill sites in the Gulf of Mexico and Caribbean Sea. Proc. 1981 Oil Spill Conf. EPA, API, USGS, Atlanta, Ga.

Gundlach, E.R., G.I. Scott and W.P. Davis. 1979. Preliminary assessment of the Howard Star oil spill site, Tampa Bay, Florida. Report to Regional Response Team. 58 p.

Gundlach, E.R., G.I. Scott, M.O. Hayes, C.D. Getter and W.P. Davis. 1979. Ecological assessment of the Peck Slip oil spill in eastern Puerto Rico, p. 303. In Proc. Ecological Damage Assessment Conf., Soc. of Petroleum Industry Biologists.

Harbison, P. 1986. Mangrove muds - a sink and a source for trace metals. Mar. Poll. Bull. 17:246-250.

Hardy, J.T., E.A. Crecelius, C.W. Apts, and J.M. Gurtisen. 1987. Sea- surface contamination in Puget Sound: Part II. Concentration and distribution of contaminants. Mar. Environ. Res. 23:251-257.

Hartung, R. and G.W. Klingler. 1970. Concentration of DDT by sedimented polluting oil. Environmental Science and Technology 4(5):407-410.

Harvey, G.R., W.G. Steinhauer, and J.M. Teal. 1972. Polychlorobiphenyls in North Atlantic Ocean water. Science 186:643-644.

Hayes, M.O. and E.R. Gundlach. 1979. Coastal processes field manual for oil spill assessment. Prepared for NOAA, Office of Marine Pollution Assessment, Boulder, Colorado. Research Planning Institute, Inc., South Carolina.

Hayes, M.O., E.R. Gundlach and C.D. Getter. 1980. Sensitivity ranking of energy port shorelines. Proc. Energy Ports Conference, ASCE, Norfolk, Va. Preprint. 13 p.

Jagtap, T.G. and A.G. Untawale. 1980. Effect of petroleum products on mangrove seedlings. Mahasagar Bulletin of the National Institute of Oceanography 13(2):165-172.

Levings, S.C., S.D. Garrity and K.A. Burns. 1994. The Galeta oil spill. III. Chronic reoiling, long-term toxicity of hydrocarbon residues and effects on epibiota in the mangrove fringe. Estuar. Coast. Shelf Sci. 38:365-395.

Lewis, R.R., III. 1979. Oil and mangrove forests: the aftermath of the Howard Star oil spill. Florida Scientist 42(Suppl.):26.

Lewis, R.R., III. 1980. The impact of the Howard Star oil spill on marine communities in Tampa Bay, Florida, vol. I. Report submitted to the Hillsborough County Board of Commissioners. Mangrove Systems, Inc., Tampa.

Lugo, A.E., G. Cintron and C. Goenaga. 1981. Mangrove ecosystems under stress, p. 129-153. In G.W. Barrett and R. Rosenberg (eds.) Stress Effects on Natural Ecosystems. John Wiley & Sons Ltd., Great Britain. 305 p.

Mackin, P. 1950. Effects of crude oil and bleedwater on oysters and aquatic plants. Texas A&M Research Foundation. Manuscript.

McMillan, C. 1974. Salt tolerance of mangroves and submerged aquatic plants, p. 379-390. In R.J. Reimold and W.H. Queen (eds.) Ecology of Halophytes. Academic Press, New York.

Odum, W.E. and R.E. Johannes. 1975. The response of mangroves to man-induced environmental stress, p. 52-62. In E.J. Ferguson-Wood and R.E. Johannes (eds.) Tropical Marine Pollution. Elsevier Oceanography series no. 12. Elsevier Scientific Publ. Co., Amsterdam.

Page, D.S., D.W. Mayo, J.F. Cooley, and E. Sorenson. 1979. Hydrocarbon distribution and weathering characteristics at a tropical oil spill site, p. 709-712. Proc. of the 1979 Oil Spill Conf.: Prevention, Behavior Control, Cleanup. EPA/API/USGS. American Petroleum Institute Publ. 4334.

Peterson, P.J., M.A.S. Burton, M. Gregson, S.M. Nye and E.K. Porter. 1979. Accumulation of tin by mangrove species in West Malaysia. The Science of the Total Environment 11:213-221.

Quilici, A., C. Infante, J. Rodreguez-Grau, J.A. LaSchiazza, H. BriceFlo and N. Pereira. 1995. Mitigation strategies at an estuarine mangrove area affected by an oil spill. Proc. of the 1995 Oil Spill Conf.: Achieving and Maintaining Preparedness. EPA/API/USCG,IMO IPIECA.

Rielinger, D.M. 1991. Respiration in black mangrove (*Avicennia germinans* [L.] Stearn. pneumatophores under submerged and oiled conditions. M.S. Thesis. University of Miami, Coral Gables. 71 p.

Seba, D.B. and E.F. Corcoran. 1969. Surface slicks as concentrators of pesticides in the marine environment. Pestic. Monit. J. 3:190-193.

Snedaker, S.C. 1995. Mangroves and climate change: scenarios and hypotheses. Hydrobiologia 295:43-49.

Snedaker, S.C., S.J. Baquer, P.J. Behr and S.I. Ahmed. 1994. Biomass distribution in *Avicennia marina* plants in the Indus River delta, Pakistan. Proceedings Pak-US Conference on Arabian Sea Living Resources and the Environment, 20-24 June 1993. Karachi, Pakistan. [in press]

Snedaker, S.C. and M.S. Brown. 1981. Water quality and mangrove ecosystem dynamics. EPA-600/4-81-022. Prepared for Office of Pesticides and Toxic Substances. U.S. Environmental Protection Agency. Environmental Protection Agency. Environmental Research Laboratory, Gulf Breeze, Florida. 80 p.

Snedaker, S.C., J.A. Jimenez and M.S. Brown. 1981. Anomalous aerial roots in *Avicennia germinans* (L.) L. in Florida and Costa Rica. Bull. Mar. Sci. 3(2):467-470.

Teal, J.M., J.W. Farrington, K.A. Burns, J.J. Stegeman, B.W. Tripp, B. Woodin and C. Phinney. 1992. The West Falmouth oil spill after twenty years; fate of oil compounds and effects on animals. Mar. Poll. Bull. 24:607-614.

Teas, H.J., A.H. Lasday, L.E. Luque, R.A. Morales, M.E. de Diego and J.E. Baker. 1989. Mangrove restoration after the 1986 Refinera Panama oil spill. Proceedings of the 1989 Oil Spill Conference. API/EPA/USCG. Washington, D.C., pp. 433-437.

Teas, H.J, R.R. Lessard, G.P. Canevari, C.D. Brown and R. Glenn. 1993. Saving oiled mangroves using a new non-dispering cleaner. Proceedings of the 1993 Oil Spill Conference: Cleanup Operations. Washington, D.C., pp. 147-151.

Thomas, C. 1987. Uptake and effects of petroleum hydrocarbons on *Rhizophora mangle* L. M.S. Thesis. University of Miami, Coral Gables. 175 p.

Thomas, C. and Ong Jin-Eong. 1981. Effect of heavy metal zinc and lead on *Rhizophora mucronata* Lam. and *Avicennia alba* Bl. seedlings. In E. Soepadmo (ed.) Proceedings of the Asian Mangrove Symposium, 25-29 Aug. 1980, Kuala Lumpur, Malaysia.

Tomlinson, P.B. 1986. The Botany of Mangroves. Cambridge University Press. Cambridge. 413 p.

Tripp, M. and R.C. Harriss. 1975. Role of mangrove vegetation in mercury cycling in the Florida Everglades, pp. 489-497. In J.O. Nriagu (ed.) Environmental Biogeochemistry, vol. 2. Metals Transfer and Ecological Mass Balances. Ann Arbor Science Publ., Inc., Ann Arbor, Michigan.

von Westernhagen, H., M. Landolt, R. Kocan, G. Furstengerg, D. Janssen, and K. Kremling. 1987. Toxicity of sea-surface microlayer: Effects on herring and turbot embryos. Mar. Environ. Res. 23:273-290.

Walsh, G.E., K.A. Ainsworth and R. Rigby. 1979. Resistance of red mangrove *(Rhizophora mangle* L.) seedlings to lead, cadmium, and mercury. Biotropica 11(1):22-27.

Wanless, H.R., 1982. Sea level is rising-so what? Jour. Sed. Petrology 52(4):1051-1054.

APPENDIX A
SUMMARY OF EXPERIMENTAL EFFECTS OF PETROLEUM ON MANGROVES

Thomas, C. 1987. Uptake and effects of petroleum hydrocarbons on *Rhizophora mangle* L. M.S. Thesis. University of Miami, Coral Gables. 175 p.

Oil Types: Mineral oil (MO),
No. 50 lubricating oil (LO)
Arabian light crude oil (CO)

Treatment Concentrations in g m^{-2}: 0 (control), 100, 375 and 650
Experiment duration: 12 weeks
All effects expressed relative to controls
Statistically insignificant effects ($p > 0.5$) are not listed
* denotes unanticipated effect

SUMMARY OF RESULTS

STEM HEIGHT GROWTH

Effect of oil type:
* MO stimulated growth 13.7%
LO inhibited growth 47.9-118.8% within 3 weeks
CO inhibited growth 47.0-88.0% after 8 weeks

Effect of oil concentration:
* MO-650 stimulated growth 13.7%
LO-375 reduced growth 47.9% at end of 12 weeks
LO-650 reduced growth 118.8% at end of 12 weeks
CO-375 reduced growth 88.0% at end of 12 weeks
CO-650 reduced growth 47.0% at end of 12 weeks

STEM DIAMETER GROWTH

Effect of oil type:
LO inhibited growth 75% after 2 weeks
* CO stimulated growth 95.7% at end of 12 weeks

Effect of oil concentration:
LO-650 inhibited growth 75.0% after 2 weeks
* CO-650 stimulated growth 95.7% at end of 12 weeks

PRIMARY BRANCH DEVELOPMENT

Effect of oil type:
* MO stimulated development 20-70%
LO inhibited development 60-70%
CO inhibited development 30-50%

Effect of oil concentration:
* MO-100 stimulated development 20%
* MO-375 stimulated development 70%
* MO-650 stimulated development 20%
LO-100 inhibited development 70%
LO-375 inhibited development 60%
CO-100 inhibited development 30%
CO-375 inhibited development 50%

SECONDARY BRANCH DEVELOPMENT
Effect of oil type:
MO caused variable effects
LO caused variable effects
* CO stimulated development 2x to 5x
Effect of oil concentration:
* MO-100 stimulated development 8x
* MO-375 stimulated development 5x
MO-650 inhibited development - no new secondary branches
LO-100 inhibited development - no new secondary branches
* LO-375 stimulated development 2x
LO-650 inhibited development - no new secondary branches
* CO-100 stimulated development 2x
* CO-375 stimulated development 3.5x
* CO-650 stimulated development 5x

NUMBER OF LEAF BUDS
Effect of oil type:
MO inhibited initiation 93.8%
LO inhibited initiation 50.0-93.8%
CO inhibited initiation 37.5-87.5%

Effect of oil concentration:
MO-650 inhibited initiation 93.8%
LO-100 inhibited initiation 93.8%
LO-375 inhibited initiation 50.0%
LO-650 inhibited initiation 18.8%
CO-100 inhibited initiation 87.5%
CO-375 inhibited initiation 37.5%

NUMBER OF JUVENILE LEAVES
Effect of oil type:
LO altered timing of leaf formation
CO altered timing of leaf formation
Effect of oil concentration:
LO variability proportional to time following exposure
CO variability proportional to time following exposure

NUMBER OF MATURE LEAVES
Effect of oil type:
* MO stimulated growth 25.0-37.5%
LO caused variable effects
CO inhibited growth 12.5-75.0%

Effect of oil concentration:
* MO-100 stimulated growth 37.5%
* MO-375 stimulated growth 25%
LO-100 inhibited growth 62.5%
LO-375 inhibited growth 37.5%
* LO-650 stimulated growth 37.5%
CO-100 inhibited growth 75%
CO-375 inhibited growth 12.5%
CO-650 inhibited growth 12.5%

LEAF HERBIVORY
Effect of oil type:
MO reduced herbivory
LO reduced herbivory
CO caused variable effects
Effect of oil concentration:
MO-375 inhibited herbivory 3x
MO-650 inhibited herbivory 50%
LO-375 inhibited herbivory 1.5x
LO-650 inhibited herbivory 1.5x
* CO-375 stimulated herbivory 4x
CO-650 inhibited herbivory 90%

LEAF DEFORMATION
Effect of oil type:
* LO reduced leaf deformation
* CO reduced leaf deformation
Effect of oil concentration:
* LO-100 reduced number of deformed leaves 2x
* LO-375 reduced number of deformed leaves 2x

* CO-100 reduced number of deformed leaves 2x
* CO-375 reduced number of deformed leaves 2x

LEAF LENGTH AT MATURITY

Effect of oil type:
MO reduced leaf length 1x to 5x
LO caused variable effects
CO reduced leaf length 1x to 6.5x

Effect of oil concentration:
MO-100 reduced leaf length 5x
MO-375 reduced leaf length 3.5x
MO-650 reduced leaf length 1.5x
* LO-100 increased leaf length 3.5x
LO-375 reduced leaf length 2.5x
LO-650 reduced leaf length 6.5x

LEAF WIDTH AT MATURITY

Effect of oil type:
MO reduced leaf width 1.8x to 3x
LO caused variable effects
CO reduced leaf length 1.5x to 3.5x
Effect of oil concentration:
MO-100 reduced leaf width 3x
MO-375 reduced leaf width 2x
MO-650 reduced leaf width 1.8x
* LO-100 increased leaf width 2.5x
LO-375 reduced leaf width 2x
LO-650 reduced leaf width 3x

NUMBER OF CHLOROTIC LEAVES

Effect of oil concentration:
MO-375 2x increase between weeks 2 and 5
LO-375 5x increase during first 5 weeks
CO-100 2x increase during first 5 weeks
CO-375 3x increase during first 5 weeks
CO-650 100% increase during first 5 weeks

NUMBER OF SENESCENT LEAVES AT END OF EXPERIMENT

Effect of oil type and concentration:
MO more senescent leaves 3x to 5x
LO more senescent leaves 2x to 7x
CO more senescent leaves 2x to 7x

DEFOLIATION DURING EXPERIMENT
Effect of oil type and concentration:
    MO between weeks 5 and 12
    LO between weeks 3 and 5
    CO first week, and between weeks 7 and 11

ORDER OF EFFECTS DUE TO OIL TYPE:
    Mineral Oil < Lubricating Oil < Crude Oil

ORDER OF EFFECTS DUE TO CONCENTRATION
    MO-100 < MO-375 and MO-750
    LO-100 < LO-375 > LO-650
    CO-100 < CO-375 > CO-650

R. mangle synthesizes a range of biogenic aliphatic hydrocarbons from C14 to C29 dominated by odd-number compounds. The highest concentrations occur among C23, C25 and C27-29 compounds. Natural synthesis and presence of aromatic compounds is low. The uptake of aromatic compounds from oil reflects the aromatic composition of the treatment oil and is inversely proportional to the molecular weight of individual aromatic compounds.

APPENDIX B
EFFECTS ENDPOINTS

The effects endpoints are derived from the following suite of morphological parameters developed from observations of the responses of mangroves to petrogenic compounds (Odum and Johannes 1975; Chan 1977; Page et al. 1979; Gundlach et al. 1979a,b; Lewis, 1979; Getter et al. 1980a,b; Hayes et al. 1980; Getter et al. 1980; Jagtap and Untawale 1980; Lugo, Cintron and Goenaga 1981; Snedaker, Jimenez and Brown 1981; Getter 1983; Ballou et al. 1987; Thomas 1987; Rielinger 1991; Quilici et al. 1995). These morphological effects endpoints are supplemented with physiological measurements including, but not limited to, leaf water potentials and chlorophyll fluorescence induction (Katusky effect). An * identifies those parameters best suited for use in shortterm experiments.

Summary of Responses of mangroves to petrogenic stress

Foliage and Canopy
    reduced leaf number per branch
    *reduced leaf size, twisting or curling
    *increased variability in leaf size and shape
    *altered leaf maturation sequences

change in leafing and shedding patterns
*abscission of buds and immature leaves
*spotty chlorosis or necrosis
reduced leaf area index

Reproductive Structures
absent or grossly excessive flowering
change in timing of flowering or fruit set
developmental failure of fruit
*abortion of flowers or immature fruit
deformed seeds or propagules
*failure to change floating orientation

Regeneration
*failure to establish primary root system
*failure to initiate primary branching
*failure in geotropic orientation in propagules
*abnormal growth forms in young seedlings
*chlorosis or necrosis of propagules

Trunks and Branches
top-dying and lowering of canopy height
mortality in outer-most sun branches
*cessation of terminal shoot growth
fissuring and cracking of bark
*expanded or more numerous lenticels
*shortened internode distances
appearance of trunk sprouts

Aerial Roots Structures
proliferation of undersized prop roots
twisting or curling of pneumatophores
*presence of adventitious aerial roots
*death of prop root tips
fissuring or pealing of periderm
*abnormal branching of prop root tips

Gross Physiology
*abnormal increase or decrease in osymolytes
*increased stomatal resistance - decreased stomatal conductance
*reduced transpiration and carbon dioxide uptake
*delayed chlorophyll activitation in response to light
*abnormal increase or decrease in respiration
*reduced rate of sap flow in primary trunk

*increased salt concentration in soft tissues, e.g., leaves
*increased tissue concentration of abscisic acid

APPENDIX B LITERATURE CITED

Ballou, T.G., R.E. Dodge, S.C. Hess, A.H. Knap and T.D. Sleeter. 1987. Effects of a dispersed and undispersed crude oil on mangroves, seagrasses and corals. Report for American Petroleum Institute. Planning Research Institute Inc., Columbia, South Carolina and Bermuda Biological Station for Research, Bermuda.

Chan, I.E. 1977. Oil pollution and tropical littoral communities: biological effects of the 1975 Florida Keys oil spill, p. 539-542. In J.O. Ludwigson (ed.) Proc. of the 1977 Oil Spill Conference. American Petroleum Institute, U.S. Environmental Protection Agency and U.S. Coast Guard, API Publ. No. 4284. 640 p.

Getter, C.D. 1983. A laboratory determination of the sensitivity of mangrove seedlings to oil and dispersant combinations. Report submitted to the Exxon Oil Spill Chemicals, Fate and Effects Environmental Subcommittee, RPI/R/82/10/28-30. Research Planning Institute, Inc., Columbia, South Carolina.

Getter, C.D., S.C. Snedaker and M.S. Brown . 1980. Assessment of biological damages at the Howard Star oil spill site, Hillsborough Bay and Tampa Bay, Florida. Report. Florida Department of Natural Resources, Tallahassee. 65 p.

Getter, C.D. J. Michel, E.R. Gundlach and G.I. Scott. 1980a. Biological changes of mangrove and sand beach communities at the Peck Slip oil spill site, eastern Puerto Rico. Report. Office Marine Pollution Assessment. NOAA. 63 p.

Getter, C.D., J.M. Nussman, E.R. Gundlach and G.I Scott. 1980b. Biological changes of mangroves and sand beach communities at the Peck Slip oil spill site, eastern Puerto Rico. RPI Report to NOAA. RPI/R/80/2/18-6. Office of Marine Pollution Assessment, Boulder, Colorado. Research Planning Institute, Inc., Columbia, South Carolina. 63 p.

Gundlach, E.R., G.I. Scott and W.P. Davis. 1979. Preliminary assessment of the Howard Star oil spill site, Tampa Bay, Florida. Report to Regional Response Team. 58 p.

Gundlach, E.R., G.I. Scott, M.O. Hayes, C.D. Getter and W.P. Davis. 1979. Ecological assessment of the Peck Slip oil spill in eastern Puerto Rico, p. 303. In Proc. Ecological Damage Assessment Conf., Soc. of Petroleum Industry Biologists.

Hayes, M.O., E.R. Gundlach and C.D. Getter. 1980. Sensitivity ranking of energy port shorelines. Proc. Energy Ports Conference, ASCE, Norfolk, Va. Preprint. 13 p.

Jagtap, T.G. and A.G. Untawale. 1980. Effect of petroleum products on mangrove seedlings. Mahasagar Bulletin of the National Institute of Oceanography 13(2):165-172.

Lewis, R.R., III. 1979. Oil and mangrove forests: the aftermath of the Howard Star oil spill. Florida Scientist 42(Suppl.):26.

Lugo, A.E., G. Cintron and C. Goenaga. 1981. Mangrove ecosystems under stress, p. 129-153. In G.W. Barrett and R. Rosenberg (eds.) Stress Effects on Natural Ecosystems. John Wiley & Sons Ltd., Great Britain. 305 p.

Odum, W.E. and R.E. Johannes. 1975. The response of mangroves to man-induced environmental stress, p. 52-62. In E.J. Ferguson-Wood and R.E. Johannes (eds.) Tropical Marine Pollution. Elsevier Oceanography series no. 12. Elsevier Scientific Publ. Co., Amsterdam.

Page, D.S., D.W. Mayo, J.F. Cooley, and E. Sorenson. 1979. Hydrocarbon distribution and weathering characteristics at a tropical oil spill site, p. 709-712. Proc. of the 1979 Oil Spill Conf.: Prevention, Behavior Control, Cleanup. EPA/API/USGS. American Petroleum Institute Publ. 4334.

Quilici, A., C. Infante, J. Rodreguez-Grau, J.A. LaSchiazza, H. BriceFlo and N. Pereira. 1995. Mitigation strategies at an estuarine mangrove area affected by an oil spill. Proc. of the 1995 Oil Spill Conf.: Achieving and Maintaining Preparedness. EPA/API/USCG,IMO IPIECA.

Rielinger, D.M. 1991. Respiration in black mangrove (*Avicennia germinans* [L.] Stearn. pneumatophores under submerged and oiled conditions. M.S. Thesis. University of Miami, Coral Gables. 71 p. Quilici, A., R. Vasquez, C. Infante, J. Rodrigues-Grau, J. LaSchiazza,

Snedaker, S.C., J.A. Jimenez and M.S. Brown. 1981. Anomalous aerial roots in *Avicennia germinans* (L.) L. in Florida and Costa Rica. Bull. Mar. Sci. 3(2):467-470.

Thomas, C. 1987. Uptake and effects of petroleum hydrocarbons on *Rhizophora mangle* L. M.S. Thesis. University of Miami, Coral Gables. 175 p.

ACKNOWLEDGMENTS

This review of oil spills and mangroves was supported in part by South Florida Oil Spill Research Center at the University of Miami, and the Florida Power & Light Company, Juno Beach, Florida.

Species Dependent on Mangroves: Mangrove Epibiota and Other Associated Species

Sally C. Levings and Stephen D. Garrity
Coastal Zone Analysis, Inc.
P.O. Box 97
Sopchoppy, FL 32308

ECOLOGICAL IMPORTANCE OF MANGROVE EPIBIOTA

Mangroves grow on the interface between the land and the sea, and are among the most productive habitats known (Lugo and Snedaker 1974). Mangroves root in soft sediments. Their stems, pneumatophores and prop roots are islands of hard substrate where plants and animals can attach and grow. These plants and animals are collectively known as the epibiota. Epibiotic assemblages can be complex, composed of diverse groups of sponges, algae, molluscs and other invertebrates (e.g. Rutzler and Feller 1987, Sutherland 1980, Farnsworth and Ellison 1996), or relatively simple, dominated by single species or a small number of species (e.g. Post 1963, Levings et al. 1994). There are strong variations in the amount, type and productivity of epibiotic assemblages dependent upon location within the forest, salinity, availability of nutrients and light, water movement and other ecological factors (e.g. Odum et al. 1982, review in Garrity and Levings 1993a). For example, epiphytic algae can contribute substantially to the primary productivity of the forest in some habitats (Burkholder and Almodovar 1973, Rodriguez and Stoner 1990), while in other areas oysters attached to red mangrove prop roots are abundant, grow quickly and form the basis of a small-scale fishery (Mattox 1949, Nikolic et al. 1976).

Mangroves also support a large and varied group of mobile organisms (e.g. Odum et al. 1982, Odum and McIvor 1990). Species vary from birds nesting in mangrove rookeries to fish living and feeding among submerged prop roots to snails feeding on attached oyster to nematodes living in mangrove sediments (Thayer et al. 1987, Lewis et al. 1985, Sheridan 1992). Oil spills into mangroves thus have the potential to affect not only mangroves, but also the species that grow attached to them and those nesting, feeding or sheltering in the mangal.

Effects of Oil Spills on Epibiota

When oil spills into a mangrove forest, epibiotic assemblages growing on mangroves are likely to be affected on two different time scales. First, soon after the spill, epibiota may be reduced either by the mechanical effects of oiling (smothering) or by the toxicity of the hydrocarbons (on direct contact or dissolved in the water column). The mechanical effects of oiling may reach mangrove surfaces from the intertidal to shallow subtidal levels, while effects of toxicity extend to the subtidal. In the long-term, epibiota may be affected by (1) the loss of surface area for attachment as the trees die and decay, (2) by physical changes in the habitat related to oiling (e.g. increases in light transmission related to defoliation, increased sediment loads caused by erosion), and (3) by toxic hydrocarbons released from sediment reservoirs. We briefly discuss each of these effects below. However, we note that data on the effects of oil spills on the epibiota are largely fragmentary or anecdotal (e.g. review in Garrity and Levings 1993a, see also Marshall et al. 1990), making generalizations difficult. Finally we briefly summarize results from the Bahia las Minas oil spill, the best studied case to date. Almost no information is available on associated mobile species; however, direct and indirect effects are likely, if undocumented.

(1) Immediate effects of oiling on epibiota

Limited data suggest extensive mortality of epibiota immediately after oil spills is common (reviewed in Garrity and Levings 1993a). Although few organisms epibiotic on mangroves have been specifically studied, laboratory testing of numerous types of similar organisms suggests that both mechanical and toxic effects of oil are expected to kill epibiota soon after exposure to oil (reviews in NRC 1985, Capuzzo 1987, Howarth 1988). Thus although documentation of oil-related losses is often poor, it probably occurs frequently.

(2) Long-term loss of surface area for attachment of epibiota

Loss of trees or sections of forests after oiling has been widely reported (see Snedaker et al. this report); epibiota are also lost when their surface for attachment dies and rots. One additional type of loss of surface area has also been shown. When mangrove trees suffer partial mortality (loss of prop roots, pneumatophores or one of several trunks), loss of surfaces for attachment of epibiota also occurs and can be extensive (Garrity et al. 1994, Levings and Garrity 1994). Even where effects of oiling do not result in adult mortality, partial mortality may reduce the standing crop of epibiota by reduction in surface area for attachment alone.

(3) Long-term physical changes in the habitat

The types of physical changes to the habitat associated with oiling are broadly similar to natural ecological changes known to affect the distribution and abundance of plants and animals. For example, changes in the amount of light reaching mangrove surfaces (associated with partial or complete defoliation of trees, Garrity et al. 1994) will change growing conditions for photosynthetic organisms, perhaps shifting the relative growth rates of species requiring light compared with those dependent upon filter feeding. Increases in sediment loads may negatively affect filter feeders, but have lesser effects on attached algae. Although there is little data to test these possibilities, there is no reason that physical changes associated with oiling would have different effects than physical changes caused by natural mortality (e.g. lightning strikes, hurricanes, Smith et al. 1994).

(4) Toxic hydrocarbons released from sediment reservoirs over time.

Recent results suggest toxic hydrocarbons can persist for 20 years or more in salt marsh and mangrove sediments (Teal et al. 1992, Corredor et al. 1990, Burns et al. 1993a,b). Even small amounts of tissue hydrocarbons have been associated with significant, negative physiological effects (Bayne et al. 1982, Capuzzo 1987, Farrington 1988, Howarth 1988, Widdows et al. 1990, see also Southward 1982). In areas where sediments contain measurable amounts of oil, toxic hydrocarbons will probably continue to affect epibiota until sediment reservoirs are depleted. Given the long persistence of sediment hydrocarbons, this may require on the order of decades rather than years (Burns et al. 1993b).

(5) One case study: The 1986 Bahia las Minas oil spill

The best studied case to date is that of the 1986 Bahia las Minas oil spill in Panama (Burns et al. 1993a,b, Garrity and Levings 1993a, b, Burns and Yelle-Simon 1994, Garrity et al. 1994, Levings and Garrity 1994, Levings et al. 1994). In 1986, the rupture of an oil storage tank spilled an estimated 75,000 to 100,000 barrels of medium weight crude oil into Bahia las Minas on the Caribbean coast of Panama (Cubit et al. 1987). The volume of this spill was greater than that of any other oil spill near mangroves and coral reefs in the tropical Americas (Keller et al. 1993).

The red mangrove fringe in replicated oiled and unoiled sites was studied for five years in three habitats: the shoreward margins of reef flats that fronted the open sea, the edges of channels and lagoons, and the banks of streams and man-made cuts that drained interior mangrove or upland forests. After five years, the length of shoreline fringed by mangroves had decreased in oiled relative to unoiled sites; surviving fringe at oiled sites had fewer and shorter prop roots and a higher proportion of dead roots than trees at unoiled sites. The net result of these physical changes was the reduction of surface area on submerged prop roots by 33% on the open coast, 38% in channels and 74% in streams, representing a strong effect of oiling on the surface area available for epibiotic attachment. Partial defoliation at oiled sites increased the amount of light reaching submerged prop roots by ~45% on the open coast and in channels and by 166% in streams.

Prop roots in each habitat were covered with a distinct epibiotic assemblage: foliose algae and sessile invertebrates on the open coast, oysters (*Crassostrea virginica*) and other bivalve molluscs and barnacles in channels, and false mussels (*Mytilopsis sallei*) in streams. Immediately after the oil spill there were massive die-offs of plants and animals attached to submerged prop roots in all three habitats (Garrity and Levings 1993 a,b). On the open coast, foliose algae and sessile invertebrates covered significantly less space on submerged prop roots for 4-5 years post-spill. In channels and streams, bivalve molluscs (oysters in channels, false mussels in streams) were reduced in abundance through five years when observations ended. The net result of oil-induced changes in (1) the structure of the mangrove fringe, and (2) epibiotic cover five years post-spill was a 33% reduction in standing crop of sessile invertebrates and foliose algae on the open coast, 65% of oysters in channels and 99% of false mussels in streams (Levings et al. 1994). If results from this study are generally applicable, effects of oiling on epibiota will probably be strong and persistent after major oil spills.

CONCLUSIONS

Oil spills have the potential to cause severe, persistent damage to mangrove forests and the species associated with them. For one case study, information suggest losses of 33-99% of epibiota five years post-spill. Although almost no data are available to evaluate quantitatively effects on mobile species, strong and persistent effects appear likely.

ACKNOWLEDGEMENTS

Funding for the authors' work in Panama was provided through contracts between the Minerals Management Service of the U.S. Department of the Interior and the Smithsonian Institution. We thank J. J. Kendall of MMS for his continued support of our work.

BIBLIOGRAPHY

Bayne, B. L., J. Widdows, M. N. Moore, P. Salkeld, C. M. Worrall and P. Donkin. 1982. Some ecological consequences of the physiological and biochemical effects of petroleum hydrocarbons on marine molluscs. Philosophical Transactions of the Royal Society of London, Series B. 297: 219-239.

Burkholder, P. R. and L. R. Almodovar. 1973. Studies on mangrove algal communities in Puerto Rico. Florida Scientist 36: 66-74.

Burns, K. A., S. D. Garrity, D. Jorissen, J. MacPherson, M. Stoelting, J. Tierney and L. Yelle-Simmons. 1993. The Galeta oil spill II. Unexpected persistence of oil trapped in mangrove sediments. Estuarine Coastal and Shelf Sciences 38: 349-364.

Burns, K. A., S. D. Garrity and S. C. Levings. 1993. Review: How many years until mangrove ecosystems recover from catastrophic oil spills? Marine Pollution Bulletin 26: 239-248.

Burns, K. A. and L. Yelle-Simmons. 1993. The Galeta oil spill IV. Relationship between sediment and organism hydrocarbon loads. Estuarine Coastal and Shelf Sciences 38: 397-412.

Capuzzo, J. M. 1987. Biological effects of petroleum hydrocarbons: assessments from experimental results. pp. 343-410 in: Long-term environmental effects of offshore oil and gas development (eds. D. F. Boesch and N. N. Rabelais). Elsevier Applied Science, London.

Corredor, J. E., J. M. Morrell and C. E. Del Castillo. 1990. Persistence of spilled oil in a tropical intertidal environment. Marine Pollution Bulletin 21: 385-388.

Cubit, J. D., C. D. Getter, J. B. C. Jackson, S. D. Garrity, H. M. Caffey, R. C. Thompson, E. Weil and M. J. Marshall. 1987. An oil spill affecting coral reefs and mangroves on the Caribbean coast of Panama, Proceedings of the 1987 Oil Spill Conference, pp. 401-406. API/EPA/USCG, Washington D. C.

Farnsworth, E. J. and A. M. Ellison. 1996. Scale-dependent spatial and temporal variability in biogeography of mangrove epibiont communities. Ecological Monographs 66: 45-66.

Farrington, J. W. 1988. Bioaccumulation of hydrophobic organic pollutant compounds. pp. 279-313 in: Ecotoxicology: Problems and approaches (eds. S. A. Levin, M. A. Harwell, J. R. Kelly and K. D. Kimball). Springer Verlag, NY.

Garrity, S. D. and S. C. Levings. 1993a. Patterns of damage and recovery from a major oil spill: the mangrove fringe and the epibiota of mangrove roots. pp. 535-792 in: Long-term assessment of the oil spill at Bahía las Minas, Panama, synthesis report, volume II: technical report (eds. B. D. Keller and J. B. C. Jackson), OCS Study MMS 93-0048. US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans LA.

Garrity, S. D. and S. C. Levings. 1993b. Effects of an oil spill on some organisms living on mangrove (*Rhizophora mangle* L.) roots in low wave energy habitats in Caribbean Panama. Marine Environmental Research 35: 251-271.

Garrity, S. D., S. C. Levings and K. A. Burns. 1994. The Galeta oil spill I. Long-term effects on the physical structure of the mangrove fringe. Estuarine, Coastal and Shelf Science 38: 327-348.

Howarth, R. W. 1988. Determining the ecological effects of oil pollution on marine ecosystems. pp. 69-97 in: Ecotoxicology: Problems and approaches (eds. S. A. Levin, M. A. Harwell, J. R. Kelly and K. D. Kimball). Springer Verlag, NY.

Keller, B. D., J. B. C. Jackson, J. D. Cubit, S. D. Garrity and H. M. Guzman. 1993. Introduction. pp. 1-24 in: Long-term assessment of the oil spill at Bahía las Minas, Panama, synthesis report, volume I: technical report (eds. B. D. Keller and J. B. C. Jackson), OCS Study MMS 93-0048. US Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans LA.

Levings, S. C. and S. D. Garrity. 1994. Effects of oil spills on fringing red mangroves (*Rhizophora mangle*): Losses of mobile species associated with submerged prop roots. Bulletin of Marine Science 54: 782-794.

Levings, S. C., S. D. Garrity and K. A. Burns. 1994. The Galeta oil spill III. Chronic reoiling, long-term toxicity of hydrocarbon residues and effects on epibiota in the mangrove fringe. Estuarine, Coastal and Shelf Science 38: 365-395.

Lewis, R. R. III, R. G. Gilmore Jr., D. W. Crewz and W. E. Odum. 1985. Mangrove habitat and fishery resources of Florida. pp. 281-336 in: Florida aquatic habitat and fishery resources. (ed. E. Seaman Jr.). Florida Chapter of the American Fisheries Society, Kissimmee.

Lugo, A. E. and S. C. Snedaker. 1974. The ecology of mangroves. Annual Review of Ecology and Systematics 5: 39-64.

Marshall, M. J., S. C. Snedaker and C. D. Getter. 1990. The sensitivity of south Florida environments to oil spills and dispersants. pp. 559-608 in: Synthesis of available biological, geological, chemical, socioeconomic and cultural resource information for the south Florida area (ed. N. W. Phillips and K. S. Larson). MMS OCS 90-0019, Minerals Management Service, U. S. Department of the Interior, Atlantic OCS Region.

Mattox, N. T. 1949. Studies of the edible oyster *Ostrea rhizophorae* Guilding in Puerto Rico. Ecological Monographs 19: 339-356.

National Research Council. 1985. Oil in the sea: inputs, fates, and effects. National Academy Press. Washington, D. C.

Nikolic, M., A. Bosch and S. Alfonso. 1976. A system for farming the mangrove oyster (*Crassostrea rhizophorae* Guilding, 1828). Aquaculture 9: 1-18.

Odum, W. E., C. C. McIvor and T. J. Smith III. 1982. The ecology of mangroves of South Florida: a community profile. FWS/OBS-81/24. U. S. Fish and Wildlife Service, Office of Biological Services.

Odum, W. E. and C. C. McIvor. 1990. Mangroves. pp. 517-548 in: Ecosystems of Florida (eds. R. L. Myers and J. J. Ewel). University of Central Florida Press, Orlando.

Post, E. 1936. Systematische und pflanzen-Geographische Notizen zur *Bostrychia-Caloglossa* Assoziation. Rev. Algol. 9: 1-84.

Rodriguez, C. and A. W. Stoner. 1990. The epiphyte community of mangrove roots in a tropical estuary: distribution and biomass. Aquatic Botany 36: 117-126.

Rützler, K. and C. Feller. 1987. Mangrove swamp communities. Oceanus 30(4): 16-24.

Sheridan, P. F. 1992. Comparative habitat utilization by estuarine macrofauna within the mangrove ecosystem of Rookery Bay, Florida. Bulletin of Marine Science 50: 21-39.

Smith, T. J. III, M. B. Robblee, H. R. Wanless and T. W. Doyle. 1994. Mangroves, hurricanes, and lightning strikes. Bioscience 44: 256-262.

Southward, A. J. 1982. An ecologist's view of the observed physiological and biochemical effects of petroleum compounds on marine organisms and ecosystems. Philosophical Transactions of the Royal Society, Series B 297: 241-255.

Sutherland, J. P. 1980. Dynamics of the epibenthic community on roots of the mangrove *Rhizophora mangle*, at Bahia de Buche, Venezuela. Marine Biology 58: 75-84.

Teal, J. M., J. W. Farrington, K. A. Burns, J. J. Stegeman, B. W. Tripp, B. Woodin and C. Phinney. 1992. The West Falmouth oil spill after 20 years: fate of fuel oil compounds and effects on animals. Marine Pollution Bulletin 24: 607-614.

Thayer, G. W., D. R. Colby and W. F. Hettler. 1987. Utilization of the red mangrove prop root habitat by fishes in South Florida. Marine Ecology Progress Series 35: 25-38.

Widdows, J., K. A. Burns, N. R. Menon, D. S. Page and S. Soria. 1990. Measurement of physiological energetics (scope for growth) and chemical contaminants in mussels (*Arca zebra*) transplanted along a contamination gradient in Bermuda. Journal of Experimental Marine Biology and Ecology 138: 99-117.

Genetic Consequences of Oil

Edward J. Klekowski
Biology Department
University of Massachuetts
Amherst, MA 01003

The long term mutational impact of oil on the plants and animals of oil spill sites is unknown. Although laboratory studies using genetically modified bacterial and animal test organisms have documented the mutagenicity of polycyclic aromatic hydrocarbons, whether such pollutant genetically degrade indigenous plants and animals is an important, but as yet unanswered, question. Key to answering this question is the development of *in-situ* forward mutation bioassays based upon key species in the native biota (both plant and animal). With such *in-situ* bioassays, the question concerning increased mutation rates in oil polluted habitats could be readily answered.

In-situ mutagen bioasays must fulfill a number of criteria in order to be useful; these include:

1. Bioassays should measure forward mutation (*i.e.* wildtype → mutant allele). The majority of laboratory assays measure back mutation (*i.e.* mutant → wildtype allele). The development of a forward mutation assay allows its implementation on wildtype organisms rather than specially constructed genotypes.

2. Bioassays should be based upon very broad genetic endpoints. A broad genetic endpoint is one that measures mutation over a set of many gene loci rather than a single locus. For example, if mutation rates are measured across a defined set of 100 gene loci, the sample size required to measure increases in mutation rate is much smaller than if the genetic endpoint is a single gene locus.

3. Many potentially mutagenic compounds (promutagens) only become active mutagens after a metabolic modification by the target organism (mutagen activation). For example, in mammals, the activation of polycyclic aromatic hydrocarbons to mutagens is catalyzed initially by NAPH-dependent cytochrome P-450 reductase. Thus for any *in-situ* bioassay, the range of metabolic pathways available that can activate different classes of promutagens should be known.

4. Mutations are the result of any one of a variety of molecular changes to the bioassay's DNA. The ideal bioassay would allow for the quantification of the kinds of molecular changes that have been induced; in this way the broad implications of the genetic degradations could be extrapolated to other organisms.

5. Not all mutations are the direct result of molecular changes to the DNA. Indirect acting mutagens may interfere with normal chromosome disjunction mechanisms during cell

division. Such chemicals will have profound genetic effects (aneuploidy). The majority of mutagen bioassays will not detect such mutagens.

With the above points in mind, let us return to the question of the genetic consequences of oil pollution. One *in-situ* mutation bioassay which fulfills two of the above five criteria (1 and 2) has been studied in oil contaminated environments, the red mangrove, *Rhizophora mangle*. In this organism, a positive correlation between mutation incidence and the concentration of polycyclic aromatic hydrocarbons of petrogenic origin in the underlying sediments was found. The sediments with the highest concentration of polycyclic aromatic hydrocarbons ($60 \mu g/g$) had a five-fold increase in mutation (Klekowski et al. 1994). Obviously what is needed is to repeat such studies in other oil contaminated environments. To this end, oil impacted mangrove forests need to be identified and studied, and, more important, new *in-situ* mutagen bioassay systems should be developed. With bioassays based upon other species, especially temperate and boreal ones, the question of the genetic degradation of the biota due to oil pollution could be answered on a global basis.

As far as future research, the most important, in our opinion, are projects that will lead to the development of rigorous *in-situ* mutagen bioassays. Emphasis should be placed upon developing assays based upon common animal and plant species from temperate coastal habitats.

LITERATURE CITED

Klekowski, E.J., J.E. Corredor, J.M. Morell, and C.A. Del Castillo. 1994. Petroleum pollution and mutation in mangroves. Marine Pollution Bulletin 28: 166-169.

The Utility of Ecological Models in the Assessment of Oil Impacts to Mangrove Ecosystems

Robert R. Twilley and Ronghua Chen
Department of Biology
University of Southwestern Louisiana
PO Box 42451
Lafayette, LA 70504

INTRODUCTION

Mangrove wetlands are the dominant ecosystem along the intertidal zone of tropical coastal landscapes (Twilley et al. 1996). The use and values of mangroves are linked to their ecological functions, and the nature of these functions can be attributed to the ecological processes of mangrove wetlands (Fig. 1). For example, ecological processes of mangroves including productivity, nutrient cycling, litter dynamics, succession, and sedimentation contribute to the habitat and water quality of estuarine and coastal waters (Fig. 1). It is the combination of these ecological processes in mangrove wetlands, along with the unique properties of tropical estuaries, that make these coastal margin ecosystems some of the most productive regions of the world (Twilley 1988, Twilley 1995). Mangroves produce a variety of forest products and support the productivity of economically important estuarine dependent fisheries in tropical estuaries. These characteristics lead to increased human utilization of mangrove resources that vary throughout the tropics depending on economic and cultural constraints (Fig. 1). Thus the use and value of mangroves are a combination of both the ecological processes of these ecosystems together with human exploitation. Therefore, any best management plan to provide for the sustainable utilization of these coastal ecosystems has to consider both ecological and social constraints of the region.

The utilization of mangrove resources can have negative feedback effects on the ecological processes of mangrove ecosystems (Fig. 1). These feedbacks can be indirect impacts such as rediversion of freshwater, which can cause changes in the productivity, litter export, and biogeochemistry of mangroves, among other processes. Other human impacts can be more direct such as the introduction of contaminants that can disrupt the natural operations of the ecosystem. Oil spills represent contaminants to mangroves that can damage the succession, productivity, and nutrient cycling of these coastal forested wetlands. These impacts have been well documented in ecological studies in Puerto Rico (Cintron et al. 1981), Panama (Duke and Pinzon 1993, Garrity et al. 1994, and others), and Gulf of Mexico (Getter et al. 1981). Negative feedbacks of contaminants such as oil in tropical coastal waters are the products of both oil exploration and transportation. These inputs become constraints on the processes of mangroves by limiting their ability to support ecological functions that provide for diverse uses and values in tropical coastal regions, such as fisheries and forestry. Efforts to minimize the damage of oil spills and enhance the recovery of mangroves are important to minimize the complex loss of ecological functions to the coastal zone.

Oil impacts can be described as changes in mangrove ecosystems relative to a natural or reference condition (reference site) to a degraded condition with different characteristics of structure and function (Fig. 2). The degree or distance of the degraded site from the natural site depends on the magnitude of the oil impact. The goal of ecological restoration is to return the degraded site back to either the natural condition or to some other new condition of restoration. These changes in the ecological characteristics of mangroves from a natural to a degraded condition, and then back to some restored condition are known as trajectories. And the nature of these trajectories will depend on the specific type of oil contamination, the magnitude of the impact, and the nature of the mangrove that is impacted. These conditions will also determine whether a restoration trajectory can return a degraded site back to a natural stage, or more likely to some different restored condition, which may be more cost-effective (considering both economic and ecological constraints). Trajectories are a very important concept in the ecological restoration of any ecosystem, since they include both the structural and functional characteristics of the sites and the time required to obtain a restored state or condition.

Ecological models can be used to assess the degree of oil spill damage and forecast the recovery of mangrove wetlands in different environmental settings depending specific levels of degradation. The ability of ecological models to analyze these trajectories depends on our knowledge of ecological processes that control the succession of ecosystems. Improvements in the information about the growth and reproduction of different forest species have led to improved individual-based simulation models that provide more realistic assessment of the development of forest ecosystems (Huston et al. 1988). Individual-based forest gap models simulate the establishment, diameter growth, and mortality of trees usually at an annual time step and on a plot of defined size (Shugart and West 1977). Models have been built for northern hardwood forests (Botkin et al. 1972), for Appalachian mountain forests (Shugart and West 1977), for southern wetland vegetation (Phipps 1979), for subtropical rain forests (Shugart et al. 1981, Doyle 1981), and for bottomland hardwood wetlands (Pearlstine et al. 1985). Gap models have been applied to examine the possible responses of forested systems to a variety of regional and global impacts (Solomon 1986, Pastor and Post 1988, Overpeck et al. 1990, Shugart et al. 1992).

This paper describes the development of an individual-based gap model of mangrove wetlands to describe the ecological processes that control forest development. We describe how this model may be modified to provide useful information in forecasting the recovery of mangrove wetlands to oil spills. We do not know of any published efforts to use ecological models for aiding management decisions regarding oil impact recovery operations. Jacobi and Schaeffer-Novelli (1990) developed a conceptual model of oil spill in mangroves, but no simulations were reported. We begin this paper with a classification of mangrove wetlands depending on coastal environmental settings and ecological types. This is followed by a description of the concept of mangrove succession. Finally, we describe the individual-based mangrove model (FORMAN) and how it may be applied to understanding recovery of mangroves to oil impacts.

ECOGEOMORPHIC APPROACH TO ASSESSMENT

The ecogeomorphic classification of mangroves describes the nature of geophysical processes of coastal environments that along with ecological processes account for the diversity of mangrove communities (Hedgpeth 1957, Thom 1984, Woodroffe 1992, Twilley 1995, Twilley et al. 1996). The landform characteristics of a coastal region together with environmental processes control the basic patterns in forest structure and growth (Thom 1982, 1984). The two basic types of geomorphological settings are those with terrigenous inputs compared to those located in carbonate environments (Fig. 3). Within each of these two categories, there are different type of settings depending on the nature of sediment, regional topography, and physical processes. Carbonate systems include continental margins and islands. Continental regions include those with peat and marl sediments, whereas islands can have either large or small catchments (high and low islands). The concept of environmental settings by Thom to explain mangrove processes from a geomorphological perspective is similar to the use of energy signature to describe the ecological processes of mangrove ecosystems (Twilley 1995, Twilley et al. 1996, Fig. 3). The hypothesis of our modeling program is that as the energy signature of a coastal region changes, there will be evidence of changes in energy flow and material cycling in mangrove ecosystems. This will be reflected in different biomass, productivity, forest regeneration rates, and exchange of nutrients and organic matter with coastal waters. Thus both the structure and function of mangrove ecosystems are linked to the geomorphology of coastal environments.

A combination of the physiographic and structural attributes of the forests together with local conditions of topography and hydrology form an ecological classification system (Lugo and Snedaker 1974, Fig. 4). This ecological scheme uses certain dominate environmental factors of the Caribbean Sea, such as soil resources and stressors, to classify mangrove forests into four types including riverine, fringe, basin and dwarf forests (Fig. 4). The formation and physiognomy of these forest types appear to be controlled strongly by local patterns of tides, terrestrial surface drainage, soil characteristics, and biological interactions. Along an inland transect, tidal frequency changes and together with changes in slope, may cause increase in salinity and other edaphic factors that limit the development of mangrove forest (mechanisms of mangrove zonation). Other ecological factors are associated with the biology of these systems such as crab predation on propagules and leaf litter (Robertson and Daniel 1989, Smith 1987, 1992). The effect of these ecological factors are twofold: 1) they decrease the maximum development of mangrove forests allowed by the existing environmental constraints of the geomorphological subclass; and 2) they control the relative zonation of mangrove species.

A combination of ecological types of mangroves can occur within any one geomorphological segment of mangroves described above depending on the distribution of soil resources and stressors within the environmental setting (Fig. 4). These ecological types are based on microtopographic and biological factors that can influence the structure and function of mangroves from shoreline to more inland locations. For instance, the juxtaposition of mangroves in a lagoon to river input will influence the fertility and hydrology of these wetlands. We refer to this hierarchical classification as the ecogeomorphic typology of mangrove wetlands. Thus there are two important descriptors in classifying mangrove wetlands: position within a specific type of geomorophological setting, and location along an inland transect (topography vector) (Figs. 3 and 4). These two characteristics of mangrove wetlands will determine the choice of appropriate landscape parameters to constrain ecological models. This is important characteristic of our modeling effort since oil impact scenarios will involve changes in both shoreline and topographic directions; and predictions of the response of mangrove wetlands will depend on the mechanistic nature of these relationships.

This use of forcing functions to describe the diversity in structure and function of mangroves along tropical coastal environments can also be used to describe the response of mangroves to contamination by oil. The ecological types of mangroves such as fringe, riverine, and basin mangroves depend on the local topography of an environmental setting. This microtopography influences the fertility and presence of stressors in mangrove soil (Fig. 4). Thus zonation patterns depend on the individual response of mangrove species to resources and stressors, together with interspecific interactions along gradients in tidal frequency. The input of oil along these topographic and tidal gradients within a region is an additional soil factor that controls the response of mangrove wetlands (Fig. 4). The scenarios of oil impacts include changes in tree mortality, recruitment of seedlings into sapling stage, defoliation, species specific tree mortality, soil remineralization, and hydrocarbon toxicity. The objective of this paper is to asses the utility of the FORMAN model to forecast the ecological processes of mangroves associated with oil contamination, specifically related to trajectories of forest structure, and function. These responses will depend on the ecogeomorphic type of mangrove.

MANGROVE FOREST DEVELOPMENT

Odum (1969) defined succession as an orderly sequence of changes in a community through time. In Odum's "neo-holistic" view, succession is solely the result of modification of the physical environment by the community, and it "culminates" in a climax in which homeostasis is achieved. Others (e.g. Horn 1974, van der Valk 1981) have defined succession as merely a pattern of changes resulting from replacement processes of individual plants. Although the mechanisms inferred in this reductionist view of community change differ from those envisioned by Odum, the concepts of the climax community are remarkably similar. Stability is then described as resistance to perturbation (Horn 1974), and the achievement of stability results in a community in which change is very slow, i.e. a climax community with homeostatic properties.

Succession in mangroves has generally been equated with zonation (Davis 1940), wherein pioneer species would be found in the fringe zones, and vegetational changes more landward would recapitulate the successional sequence in terrestrial communities. Zonation in mangrove communities has variously been accounted for by a number of biological factors including salinity tolerance of individual species (e.g. Snedaker 1982), seedling dispersal patterns resulting from different sizes of mangrove propagules (Rabinowitz 1978), differential predation by grapsid crabs (Smith 1987), and interspecific competition (Ball 1980). Snedaker (1982) proposed the establishment of stable monospecific zones wherein each species is best adapted to flourish due to the interaction of physiological tolerances of species with environmental conditions. Zones of mixed species composition have been thought of primarily as transition zones or ecotones between monospecific zones and as such have been interpreted as being temporary responses to disturbance (Lugo and Snedaker 1974, Lugo 1980).

The consideration of changes in mangrove communities in a strictly spatial rather than temporal pattern has contributed to the lack of understanding of succession in these forested wetlands. Mangrove zones, particularly those with several species, may represent temporary states in time (Davis 1940, Chapman 1976) or may be steady state communities in response to environmental conditions (Lugo 1980). The responses of individual species to physical stress may result in community trends in succession. Thus, plants in harsh environments may tend to undergo "cyclic succession" where very few species are able to colonize available space (Connell and Slayter 1977). Changes in forest structure and species importance are limited in the tropical intertidal zone of the Western Hemisphere due to the few plants that can colonize this area. Although species composition may not change appreciably over time, the functional characteristics of the system, such as community productivity, may still develop in a predictable fashion. In mangrove communities, structural and functional characteristics of the community may respond to geomorphological processes (Thom 1967), or perhaps, on a shorter time scale, to cyclic and noncyclic environmental oscillations (Lugo 1980). The temporal scale of these changes and their importance in terms of management of these ecosystems are dependent upon the relative interaction of environmental settings with biotic components of the community.

THE FORMAN MODEL

Compared with modeling of other wetland ecosystems, ecological modeling of mangroves is very primitive to nonexistent (Mitsch et al. 1988), although these wetlands are dominant in most of the tropical and subtropical coastal regions of the world. Miller (1972) developed a process model with a series of equations that predict primary productivity and transpiration of the mangrove forest canopy from input data on canopy structure and meteorological variables. The model determined that the most important variables on productivity were air temperature and humidity; and productivity decreased with increases in either of these variables. Lugo and Snedaker (1974) developed a qualitative component model to show energy and material flow, together with potential stressors that influence mangrove growth. They found the time required to reach steady state levels of mangrove biomass was similar to the average frequency of tropical hurricane, about 20 yrs. The model also demonstrated the linkage in tidal flushing and storage of detritus on the forest floor, which affected nutrient availability and productivity of the forest. Reductions in upland runoff caused a decrease in the productivity of the mangrove forest. A more complex model was developed by Odum et al. (1977) which demonstrated the importance of freshwater sources on production and tidal exchange. This model added salinity and potential evapotranspiration as forcing functions that affected mangrove productivity. The impacts of tropical storm, herbicides and nutrient enrichment on mangrove ecosystems were simulated by Sell (1977). The model suggested that complete mangrove recolonization of areas sprayed with herbicides may take 55 yrs to more than 100 yrs.

None of the present mangrove models have used the individual-based approach to understanding the fate of individual trees in a landscape. Gap models developed from temperate terrestrial forest share the same basic characteristic equation for calculating growth, influences of limiting resource availability, establishment, and mortality in a forest stand. Most of these models give abstract consideration to the growth-limiting effects of critical plant nutrients (Shugart et al. 1980). Nutrient cycling in open mangrove ecosystem is more complex than in terrestrial ecosystem, because it is controlled by both ecological feedback, and net balance of input and output operated by hydrology (Fig. 5). To simulate ecosystem development in such 'open systems', using the interaction between vegetation and available nutrients requires an understanding of allogenic and autogenic nutrient dynamics. But such synoptic investigations of nutrient process are few. We have been developing an individual-based ecosystem model to asses the development of mangroves as constrained by different edaphic conditions.

Model Design and Parameters

A flow diagram for the mangrove succession model is given in Fig. 5 and representative structure of the STELLA model is described in Fig. 6, which is a submodel for the species *Rhizophora mangle*. This model is parameterized with information on a basin mangrove forest in Rookery Bay, which is equivalent to a lagoon-basin wetland in Fig. 3. The objective of our succession model is to represent the change in species composition and biomass in a mangrove forest over time. The approach is to represent how tree growth both responds to and changes the external environment over time. Simulator tree species are defined by a few general characteristics: a maximum age; maximum diameter; maximum height; a relation between height and diameter; between total leaf weight and diameter; between rate of photosynthesis and available light; between relative growth and a measure of climate; a range of soil conditions within which the species can grow; and the number of saplings which can enter the stand. Direct competition among individuals is restricted to competition for light. Species strategies are defined by species-specific survival probabilities and by differential addition of new saplings in relation to light at the forest floor. Because the annual probability of survival of an individual is related to the maximum known lifetime of its species, individuals of long-lived species have a better chance of survival in any one year than those with short maximum lives.

The model represents tree growth on an annual basis for each size class (5). Light, temperature, salinity and space competition are all factors that reduce the growth of mangroves in a multiplicative fashion below over that attained by a tree growing under optimum conditions. The model consists of a basic growth rate equation for each species that may be taken to represent the rate of growth of a tree with optimum site quality and no competition from other trees. For each plot-year, this growth rate is decreased by factors taking into account shading and shade tolerance, soil quality (soil salinity) and average climate as measured by the number of growing degree days.

A tree growing in the open collects an amount of radiant energy roughly proportional to its leaf area. The JABOWA growth rate equation for a tree growing under optimum conditions has the form:

$$d[D^2H]/dt = R \cdot LA \cdot (1-DH/D_{max}H_{max}) \qquad (1)$$

in which D is the DBH of the tree, H is height, with D_{max} and H_{max} being maximum values of these quantities for a given species, LA is the leaf area, and R is a growth rate parameter. The equation states that the change in volume (D^2H) of a tree over a period of 1 year is proportional to the amount of sunlight which the tree receives, derated by a factor ($1-DH/D_{max}H_{max}$) which takes into account the energy required to maintain the living tissue.

Equation (1) is modified by relations between height and diameter, and basal area and leaf weight. Upon substitution and differentiation, equation 1 becomes:

$$dD/dt = GD(1-DH/D_{max}H_{max})/(274+3b_2D-4b_3D^2) \qquad (2)$$

with G (a growth parameter) being used for the product of constants, aR. Botkin et al. (1972) solved Equation (2) for G. Equation (2) is used in almost all of the currently published forest gap models as the optimal growth equation.

Given a basic equation for the growth of an individual tree, gap models incorporate the interaction of trees with one another and with their environment by modifying the growth that is predicted by the species-specific optimal growth equation. These are the equations that provide the population biology interactions into the mangrove succession model. One important consideration is the shading relation among trees; it is in computing these interactions that the spatial element of gap models arise. The light that reaches a given tree is calculated by attenuating the incident radiation by the sum of leaf areas for all trees taller than the tree:

$$Q(h) = Q_0\exp(-k \int LA(h')dh') \qquad (3)$$

where LA(h') is the distribution of leaf area as a function of height, Q_0 is the incident radiation, Q(h) is the radiation at height (h), and k is a constant. We used $k = 0.35$ in the present simulation. The shading of each tree is denoted (AL) (available light), and the reduction of growth associated with this shading is denoted r(AL). Botkin et al. (1972) used two expressions for the function, r:

$$r_s(AL) = 1-\exp(1-4.66(AL-0.05)) \qquad (4)$$

$$r_i(AL)=2.24(1-\exp(-1.136(AL-0.08))) \qquad (5)$$

where r_s and r_i are the reduction of photosynthesis rates for shade-tolerant and shade-intolerant trees, respectively.

The function for the effect of temperature on growth assumes that each species grows in response to an annual accrued number of degree days above some threshold temperature. But mangroves are tropical plants, which have a optimal growth in tropical zone with a higher biomass in lower latitude (Twilley et al. 1992). The equation used for temperature effect in the JOBOWA model was not suitable to describe the effect of climate (global temperatures) on mangrove growth. We used the following assumed equation instead to describe climate effects on mangrove growth:

$$T(DEGD) = 1-(DEGD_{min}/DEGD)^2. \qquad (6)$$

Competition for moisture and nutrients is represented by a crowding factor, which is simply the fraction of the maximum possible basal area in the 10 x10 m plot that is actually covered by trees. The function is

$$S(BAR) = 1-BASEAREA / BASEAREA_{max} \qquad (7)$$

where BASEAREA is the total basal area on the plot. $BASEAREA_{max}$ is the maximal basal area on the plot in which trees will grow. Maximum base area is 3500 $cm^2/100m^2$ in this model (Lugo et al. 1980).

Soil salinity is an important limiting factor for mangrove growth. The type of linear response model used by Mass and Hoffman (1977) (from van Genuchten and Hoffman 1984) contains two independent parameters: the salinity threshold (ct), being the maximum soil salinity without yield reduction, and the slopes (s) of the curve determining the fractional yield decline per unit increase in salinity beyond the threshold. In equation form:

$$S(SAL)=\begin{cases} 1 & 0 =< c =< ct \\ 1-s(c-ct) & < c =< c_0 \\ 0 & c>c_0 \end{cases} \qquad (8)$$

c is the average root zone salinity, c_0 is the concentration beyond which the yield is zero, 35‰ is the salinity threshold (ct) for mangrove height growth (Cintron et al. 1978).

By multiplying equation (2) by equation (6), (7), (8) and equation (4) or (5), we have the growth equation used in the FORMAN model:

$$dD/dt = [GD(1-DH/D_{max}H_{max})/(274+3b_2D-4b_3D^2)] \cdot T(DEGD) \cdot r(AL) \cdot S(BAR) \cdot S(SAL) \qquad (9)$$

Saplings are assumed to be established at a size (DBH) of 0.5 cm. We used a range of 1-3 individual sapling/yr as recruitment rate for the three mangrove species on a 10 x10 m plot. But canopy closure has a differential shade effect for shade-tolerant and shade-intolerant species.

There are two kinds of mortality in this model. One type is related to aging and the other to growth. Each tree is assumed to have an intrinsic mortality rate such that, under normal conditions, 1% of the individuals in a cohort could be expected to live long enough to attain their maximum age. Trees whose rate of growth was suppressed due to shading or other factors, to the point that their diameter increment is less than 0.01 cm per year, are subjected to additional mortality called Ps equal to 0.368, where Ps is the probability of survival of a suppressed individual. This has the effect of allowing only 1% of a suppressed cohort to survive 10 years. The rational for this function is that if a tree can not maintain a certain minimum growth rate it will be more susceptible to factor that cause mortality, such as disease.

We separated the trees into 4 cohorts (SIZE) which have 0.5-2.5 cm (sapling), 2.5-5 cm, 5-15 cm and >15 cm DBH, respectively (Fig. 6). We only modeled these 4 SIZEs because they represent the average DBH of trees of each species based on the mangrove literature for the Caribbean Sea. Equation (9) is used to calculate growth of each SIZE. The graduate rate of a cohort is divided by the average DBH width of the cohort to obtain the percent of a cohort that will graduate in any given year. Tree mortality due to both slow growth and old age are computed as fractions of trees dying rather than on probabilities, because tree numbers rather than individual trees are being modeled. We assumed that when the forest reaches its stable state, the age mortality will keep its stable mortality. It has been suggested that the structure and functional development of mangroves take about 20-30 years to achieve a stage of maturity (Lugo 1980, Warner 1990).

Biomass was calculated using the allometric equation from Cintron and Schaeffer-Novelli (1984). The average biomass of a class is multiplied by the number of trees in that SIZE and the results for all the classes are summed to obtain total biomass. The LAI of a plot is calculated by converting leaf weight to leaf area and divided by plot area.

Some preliminary simulations of succession and changes in species composition as a function of time and climate are represented by the FORMAN model in Fig. 7. Four general indicators of overall forest dynamics are the total number of stems per hectare, total basal area, mean DBH, and total above ground biomass of the stand are used as output of the model (Fig. 7). These simulations are based on propagule recruitment rate of 3-10 per 100 m^2 per yr and soil salinity of 42 ‰. Above ground biomass increases rapidly until 20 years at which time the rate of increase decreases to steady state. This decreased rate of biomass accumulation is caused by shading within the stand due to the closure of the forest canopy. After 25 years, stand biomass tends to remain relative constant. Calibrations of the model to different forest structure values at RBF1 in Rookery Bay (Warner 1990) show good agreement with FORMAN v 1.1. The different tree statistics for RBF1 are for 100 m^2 plots at increasing distances inland from the shore. The empirical data from Rookery Bay showed mangrove biomass reached 70 to 85 t/ha after about 20 yrs of disturbance by Hurricane Donna (Warner 1990).

Model Applications to Oil Impacts

Mortality: The FORMAN model (version 1.1) can be used to describe the impacts of oil on a variety of ecological processes that are important to forecast the specific recovery of mangrove wetlands. Tree mortality due both to slow growth and old age are computed as fractions of the entire cohort rather than as probabilities based on individual trees. Tree age mortality is computed by :

$$\text{gmrtage} = 1\text{-}(1\text{-kmrtage})^{\text{age}}. \tag{13}$$

Total mortality of a cohort is gmrtage times the number of trees in that specific size class (Swartzman and Kaluzny 1987). The kmrtage parameter for a species was set so that only 1 percent of the trees can reach the maximum age of the species. Because age in the FORMAN model refers to that of the forest, and not to the age of each cohort, the mortality of each cohort will increase with time. Few data are available regarding survival rate of trees. We used some assumptions, i.e. the mortality will keep its stable mortality when the forest reaches its stable state. The age of a mature stand of mangroves was based on previous research estimates (Lugo et al. 1980, Warner 1990).

An oil slick in a mangrove wetland will cause a certain mortality of trees depending on the concentration of hydrocarbons and species of trees. FORMAN can be programmed to account for the differential sensitivity of mangrove trees to specific concentrations of oil, and thus evaluate the degree of damage to the forest. Research results such as those by Duke and Pinzon (1993) and Cintron et al. (1981) give examples of the degree of tree mortality at different environmental settings, particularly soil salinity. The degree of tree mortality varies depending on the hypersalinity of the soil. Thus, those mangroves on dry coastal environments of the Gulf of Mexico and Caribbean may be more susceptible of oil spil than those in more humid environments. Also, Duke and Pinzon (1993) found that the exposure of the shore to wind and wave energy was important factor in tree mortality and growth.

Recruitment

The impacts of oil on the recruitment of mangrove seedlings into young tree cohorts can be analyzed by FORMAN (Fig. 5). Recruitment rates for each species ranges from 1-3 individuals/yr in a 100 m^2 plot. This recruitment rate can be differentially adjusted for each species based on the effects of oil type and concentration in the plot. This function of the model can also be used to test the effects of different recovery techniques that deal with replanting seedlings in stressed areas. The density of transplanting and species selection of propagules may be evaluated in FORMAN. The model can compare the projected rates of forest recovery using different rates of recruitment, given the effects of oil hydrocarbons on plant growth (see below). The simultaneous effects of oil on different ecosystem processes is the value of using ecological models to asses restoration activities and strategies.

There are several examples of replanting mangrove following oil contamination, but few studies have followed up on the efficacy of this technique compared to natural revegetation processes. Duke and Pinzon (1993) followed specific sites and determined the effect of recruitment on the recovery of mangrove forest in oiled and unoiled locations. In addition other local factors such as crab predation were considered in their study. These types of analyses are needed to parameterize the recruitment function in FORMAN. Sensitivity analyses have shown (data not shown) that the level of recruitment, and particularly species-specific rates of recruitment, can be important parameters to development of forest structure. Again, this ecological process would need to be evaluated under different levels of oil contamination and soil conditions (salinity and fertility).

Litter Fall and Nutrient Cycling

FORMAN can be used to evaluate reductions in leaf area index of a mangrove forest as a response of oil toxicity, and the fate of this additional leaf litter in the ecosystem. The present model of edaphic nutrient cycles is limited and are presently under development (Chen 1996). Thus the feedback effect of increased leaf litter on soil nutrient processes is not presently in the FORMAN model (v 1.1). However, the effect of reduced LAI on increased light and the formation of gaps in mangrove wetlands can be evaluated in the model. Oil impacts increase light resources in forests by reducing the canopy, and this will influence the relative competition of subdominants and seedlings within the gap. Again, this increase in light resources has to be evaluated simultaneously with negative effects of hydrocarbons on plant growth and recruitment. Thus an increase in this resource may have minimal effect on forest recovery and patterns of zonation following oil disturbance.

Duke and Pinzon (1993) show the sensitivity of LAI to concentrations of hydrocarbons in the soil, along with the rates of recovery of canopy structure. The rates of these processes in oiled and unoiled sites give some idea as to the magnitude of this damage as oil contamination increases. Few analyses have followed the fate of this excess organic matter from the canopy through the ecosystem.

Soil Remineralization

The present version (v 1.1) of FORMAN uses initial conditions of nutrients and salinity to simulate the mangrove growth function in the model (equation 9). Recent versions of the model include mechanistic approaches to simulate soil conditions that are linked to the forest growth model (Chen 1996). Thus the feedback effect of litter fall and soil remineralization can be evaluated in the time series development of the mangrove wetland. This is also very important to include in the evaluation of oil impacts in mangrove soils. The direct effect of hydrocarbons will be from stress to tree and seedling growth. An indirect and possibly long-term effect of oil will be on the rates of nutrient remineralization on nutrient availability, and the implications of this disturbance to the growth function. It is also important to model these soil processes to evaluate some of the bioremediation techniques suggested for mangrove restoration. These include fertilization and microbe seeding, both of which may change the fertility of forest soils. It will be important to provide these sort of feedback mechanisms in the ecological model to evaluate indirect effects of oil on forest development.

Hydrocarbon Stress

The accumulation of hydrocarbons in soil and the stress effect of this contaminant can be included in the growth function of mangroves in the FORMAN model (Fig. 5). Presently the growth function is affected by two soil factors, fertility [S(BAR)] and salinity [S(SAL)] (see equation 9). A third factor, S(OIL), could be added and reflect the effects of different concentrations of hydrocarbons on mangrove growth. This effect can be species specific and thus evaluate the differential response of mangroves to forest development. Mathematical functions describing the relative response of each mangrove species will have to be determined from the literature. Studies in Panama provide data on the potential growth of mangroves to different concentrations of hydrocarbons in soil (Duke and Pinzon 1993).

The hydrology and biological processes in mangrove soil will determine the residence time of this contaminant in the ecosystem. In addition, the accretionary processes on new soil formation will be important to determine the location of oil residues relative to the zone of active root uptake. Thus residence time is not only the decrease in concentrations of oil residue, but also the movement of this oil from zones of root impact. The present version of FORMAN does not include a hydrology model. Yet an independent hydrology model (HYMAN model) has been successfully used to simulate changes in soil salinity in basin mangrove forests in Rookery Bay, Florida. Recent model development has attempted to link hydrology and soil biogeochemistry models with forest growth models (Chen 1996).

Restoration and Remediation Strategies

Ecological modelling can be used to evaluate the probabilities of obtaining restoration goals using different strategies of rehabilitation. As described above, the FORMAN model can be used to determine the efficacy of different replanting techniques to the future development of mangrove wetlands given present soil conditions of the oil spill. The selection of species and the amount of site preparation can be included in the model to evaluate the time required to obtain forests recovery. This model can also be used to forecast the structural complexity and species composition of the forest after specific times of recovery. A very important consideration is whether the restoration goal of the restored mangrove wetland should be that of a natural or reference wetland. Damage to a site may be so severe and sensitive to oil impacts that the restored forest will not reflect the natural wetland without major site preparation and recovery. Thus the success of a restoration strategy must take into account the feasibility of obtaining the structure and function of a reference site. The FORMAN model may provide some indication of what the structural characteristics will be at the restored site at specific times following the initial restoration tasks.

Other restoration strategies include soil modifications that are expected to enhance oil degradation and export (removal) and thus stimulate tree growth. Washing oil from mangrove soils and fertilization are a couple of restoration activities that can be evaluated with FORMAN model. In the present version of the model, washing of oil would cause a reduction in the S(OIL) component of the growth model, whereas fertilization would increase the S(BAR) coefficient. Both coefficients would stimulate tree growth, depending on tree species present. These two factors could be evaluated independently and in combination for forests of different species and size class distribution. This type of sensitivity analysis of the model with changes in specific coefficients could be compared to the time required to reach a restored forest (see Fig. 2).

Another type of strategy in bioremediation includes the addition of microbial populations that would consume hydrocarbons and reduce the level of contamination in the soil. These microbes may also influence organic matter remineralization, and thus the turnover of soil nutrients. This modification is usually done with addition of nutrients, to facilitate the decomposition process (prevent microbial nutrient limitation). Parts of this process could be included in the model by changes in the S(BAR) and S(OIL) coefficients. But as described above, the mechanisms of soil remineralization processes are not included in FORMAN 1.1 and thus changes in soil conditions would not be mechanisms of the model, but known adjustments in the appropriate parameters.

There are many aspects of mangrove wetland restoration that could be addressed using sensitivity analysis of the FORMAN model. Endpoint in any ecological restoration program could be evaluated based on species richness and structural characteristics of the site, along with the length of time to attain that condition. Long vs short-term remediation goals can both be evaluated with FORMAN. Abnormal and reduced growth compared to acute and delayed mortality can be simulated and used to forecast differences in species composition and forest structure. Of course the simulated output of any ecological model is constrained by the quality of the knowledge used to formulate the response of mangrove processes to oil contamination. These simulations will also have to be compared to existing information on forest structure at specific sites. In addition, validation procedures will have to be developed before the general application of the model can be used in different coastal regions.

The FORMAN model can also be used within the framework of GIS to analyze the spatial effects of an oil spill in a coastal tropical landscape. Spatially explicit data oil contamination in sediments along with other soil characteristics such as salinity and fertility can be used to simulate responses of mangrove growth to those specific site conditions.

Model Limitations and Research Needs

There are many ecological processes associated with oil contamination in mangrove wetlands that are considered in the FORMAN (v1.1) model. Many of the applications described above are based on known relations between the amount of oil and the adjustment to some parameter that affects tree growth or mortality. Although these can prove to be very useful in evaluating different restoration strategies, they lack the mechanistic interactions that provide greater generality and precision in any ecological model. In addition, FORMAN is focused on forecasting the species richness and structural complexity of mangrove wetlands and does not include the ecology of other organisms or the habitat quality to nonresident species. Many other trophic levels and guilds in mangrove wetlands may be important to the development of mangrove trees, and need to be included in more thorough versions of the basic model. For example, FORMAN does not model directly the effects of crab predation or epibiota on the ecology of mangrove forests. Some of these effects can be included in values of parameters for mortality and recruitment, but these are then inherent and cannot be mechanistically changed during a simulation. Microbial processes are basically part of the S(BAR) parameter, and none of the nutrient remineralization processes are in this version of FORMAN. But as described previously, many of these biogeochemical processes are presently included in a recent version of the model. Also, motile fauna are not included in the model.

A research agenda should include the development of functions with several of the parameters in the FORMAN model that describe their response to specific hydrocarbon concentrations. It is not in the mission of this document to thoroughly review the present knowledge of mangrove response to oil spills to parameterize FORMAN based on several studies at oil impacted mangrove sites. This would be one of the first research priorities to develop a specific modification of the FORMAN model for the application of evaluating mangrove restoration based on the studies in Panama, Puerto Rico, Tampa, and Brazil. Once this has been accomplished, then more specific details of the research needs could be outlined. It is recommended that many of these research agendas be supported with preliminary simulation results. Sensitivity analyses of the FORMAN model with thorough determinations of parameters as function of hydrocarbon concentrations (as synthesis of existing information) could provide specific recommendations for mangrove research programs.

LITERATURE CITED

Ball, M.C. 1980. Patterns of secondary succession in a mangrove forest in south Florida. Oecologia **44**:226-235.

Botkin D. B., J. F. Janak and J. R. Wallis. 1972. Some ecological consequences of a computer model of forest growth. J. Ecol. **60**: 849-973.

Chapman, V. J. 1976. Mangrove Vegetation. J. Cramer, Germany.

Chen, R. 1996. Ecological analysis and simulation models of landscape patterns in mangrove forest development and soil characteristics along the Shark River estuary, Florida. Ph.D. Dissertation. University of Southwestern Louisiana, Lafayette.

Cintrón , G., A. E. Lugo, D. J. Pool and G. Morris. 1978. Mangroves of arid environments in Puerto Rico and adjacent islands. Biotropica **10**:110-121.

Cintrón , G., A.E. Lugo, R. Martinez, B.B. Cintron, and L. Encarnacion. 1981. Impact of oil in the tropical marine environment, pp. 18-27. Technical Publication, Division of Marine Resources, Department of Natural Resources of Puerto Rico.

Cintrón, G. and Y. Schaeffer-Novelli. 1984. Caracteristics y desarrollo estructural de los manglares de Norte y Sur America. Program Regional de Desarrollo Cientifico y Tecnologico **25**:4-15.

Connell, J. H. and R. O. Slayter. 1977. Mechanisms of succession in natural communities and their role in community stability and organization. The American Naturalist **111**:1119-1144.

Davis, J. H., Jr. 1940. The ecology and geologic role of mangroves in Florida. The Bulletin of the American Association of Petroleum Geologists **26**:307-425.

Doyle, T. W. 1981. The role of disturbance in the gap dynamics of a montane rain forest succession model. pp 56-73. *In* D. C. West, H. H. Shugart and D. B. Botkin. Eds. Forest Succession: concept and application. Springer-Verlag, New York.

Duke, N.C. and Z. Pinzon. 1993. Mangrove forests. pp 447-553. In: B.D. Keller and J.B.C. Jackson, (eds). Long-term assessment of the oil spill at Bahia las Minas, Panama, Synthesis Report, Volumn II, Technical Report. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Regional Office, New Orleans, LA.

Duke, N. C. and Z. S. Pinzon. 1992. Aging *Rhizophora* seedings from leaf scar nodes: a technique for studying recruitment and growth in mangrove forests. Biotropica **24**:173-186.

Garrity, S.D., S.C. Levings and K.A. Burns. 1994. The Galeta oil spill. I. Long-term effects on the phsical structure of the mangrove fringe. Estuarine, Coastal and shelf Science 38: 327-348.

Getter, C.D. G.I. Scott, and J. Michel. 1981. The effects of oil spills on mangrove forests: a comparison of finve oil spill sites in the Gulf of Mexico and the Caribbean Sea, pp.65-111. Proceedings of the 1981 Oil spill Conference. API/EPA/USCG, Washington, D.C.

Hedgpeth, J.W. 1957. Classification of marine environments. Geological Society of America, Memoir 67, Volume 1: 17-28.

Horn, H. S. 1974. The ecology of secondary succession. Annual Review of Ecology and Systematics **5**:25-37.

Huston, M., D. DeAngelis and W. Post. 1988. New computer models unify ecological theory. BioScience **38**: 682-691.

Jacobi, C. M. and Y. Schaeffer-Novelli. 1990. Oil spill in mangroves: a conceptual model based on long term field observation. Ecological Modelling **52**:53-59.

Lugo A. E. 1980. Mangrove ecosystems: Successional or steady state? Special edition of Biotropica: Tropical Succession 65-72.

Lugo, A. E. and S. C. Snedaker. 1974. The ecology of mangroves. Annual Review of Ecology and Systematics **5**:39-64.

Lugo, A. E., R. R. Twilley and C. Patterson-Zucca. 1980. The role of black mangrove forests in the productivity of coastal ecosystems in South Florida. Final report to U.S. EPA, Corvallis Environmental Research Laboratory, Corvallis, Oregon, Contract No. R806079010, Center for wetlands, University of Florida, Gainesville, Florida, 281 pp.

Miller, P. C. 1972. Bioclimate, leaf temperature, and primary production in red mangrove canopies in south Florida. Ecology **53**:22-45.

Mitsch, W. J., Straskraba, M. and S. E. Jørgensen. 1988. Wetland modelling. Elsevier, Amsterdam.

Odum, E. P. 1969. The strategy of ecosystem development. Science **164**:262-269.

Odum, H. T., W. M. Kemp, M. Sell, W. Boynton and M. Lehman. 1977. Energy analysis and coupling of man and estuaries. Environ. Manage. **1**: 297-315.

Overpeck, J. T., Rind, D. and R. Goldberg. 1990. Climate-induced changes in forest disturbance and vegetation. Nature **343**:51-53.

Pastor, J. and W. M. Post. 1988. Response of northern forests to CO_2-induced climate change. Nature **334**:55-58.

Pearlstine L., H. McKellar and W. Kitchens. 1985. Modelling the impacts of a river diversion on bottomland forest communities in the Santee river floodplain, South Carolina. Ecological Modelling **29**:283-302.

Phipps, R. L. 1979. Simulation of wetlands forest vegetation dynamics. Ecological Modelling **7**:257-288.

Rabinowitz, D. 1978. Early growth of mangrove seedlings in Panama, and an hypothesis concerning the relationship of dispersal and zonation. Journal of Biogeography **5**:113-133.

Robertson, A. I. and Daniel, P. A. 1989 The influence of crabs on litter processing in high intertidal mangrove forests in tropical Australia. Oecologia **78**, 191-198.

Sell, M.G., Jr. 1977. Modeling the response of mangrove ecosystems to herbicide spraying, hurricanes, nutrient enrichment and economic development. Ph.D. Thesis, University of Florida, Gainesville, Florida.

Shugart, H. H., Jr., Hopkins, M. S., Burgess, I. P. and A. T. Mortlock. 1980. The development of a succession model for subtropical rain forest and its application to assess the effects of timber harvest at Wiangaree State Forest, New South Wales. J. of Environ. Manage. **11**:243-265.

Shugart, H. H., Jr., Smith, T. and W. M. Post. 1992. Simulation models and global change. Annu. Rev. Ecol. Syst. **23**:15-38.

Shugart, H. H. and D. C. West. 1977. Development of an Appalachian deciduous forest succession model and its application to assessment of the impact of the chestnut blight. J. Environ. Manag. **5**:161-179.

Shugart, H. H., M. S. Hopkins, I. P. Burgess and A. T. Mortlock. 1981. The development of a succession model for subtropical rain forest and its application to asses the effect of timber harvest at Wiangarree State Forest, New South Wales. J. Environ. Manag. **11**:243-265.

Smith, T. J. 1987. Seed predation in relation to tree dominance and distribution in mangrove forests. Ecology **68**, 266-273.

Smith, T. J. 1992. Forest structure. In: Robertson, A. I. and Alongi, D. M. (Eds.) Tropical Mangrove Ecosystems, pp. 101-136. American Geophysical Union, Washington, D.C.

Smith, T. J., M. B. Robblee, H. R.Wanless, and T. W. Doyle. (1994) Mangroves, hurricanes, and lightning strikes. BioScience **44**, 256-262.

Snedaker, S. C. 1982. Mangrove species zonation: why?, pp. 111-125. In: D. N. Sen and K. S. Rajpurohit (eds), Tasks for Vegetation Science. Dr. W. Junk Publishers.

Solomon, A. M. 1986. Transient response of forests to CO_2-induced climate change: simulation modeling experiments in eastern North America. Hydrobiologia (Berlin) **68**:567-579.

Swartzman, G. L. and S. P. Kaluzny. 1987. Ecological simulation primer. Macmillian Publishing Company, New York. 370pp.

Thom, B. G. 1967. Mangrove ecology and deltaic geomorphology: Tabasco, Mexico. Journal of Ecology **55**:301-343.

Thom, B. G. 1982. Mangrove ecology - a geomorphological perspective, pp 3-17. In: B.F. Clough (ed.), Mangrove Ecosystems in Australia. Australian National University Press, Canberra.

Thom, B. G. 1984. Coastal landforms and geomorphic processes. pp. 3-17. In: S. C. Snedaker and J. G. Snedaker (eds.), The mangrove ecosystem: research methods. Unesco, United Kingdom.

Twilley, R. R. 1988. Coupling of mangroves to the productivity of estuarine and coastal waters, pp 155-180. In: B. O. Jansson (ed.), Coastal-Offshore Ecosystem Interactions. Springer-Verlag, Germany.

Twilley, R. R. 1995. Properties of mangrove ecosystems in relation to the energy signature of coastal environments, pp. 43-62. In: C. A. S. Hall (ed.), Maximum Power. University Press of Colorado, Niwot.

Twilley. R. R., R. H. Chen and T. Hargis. 1992. Carbon sinks in mangroves and their implications to carbon budget of tropical coastal ecosystems. Water, Air and Soil Pollution **64**:265-288.

Twilley, R. R., Snedaker, S. C., Yañez-Arancibia, A., and E. Medina. 1995. Mangrove systems, pp. 387-393. In: V. H. Heywood and R. T. Watson (eds.), Global Biodiversity Assessment. United Nations Environment Programme, Cambridge University Press, Great Britian.

Twilley, R. R., Snedaker, S. C., Yañez-Arancibia, A. and E. Medina. 1996. Biodiversity and ecosystem processes in tropical estuaries: perspectives from mangrove ecosystems. In: Mooney, H., Cushman, H. and E. Medina (eds.), Biodiversity and ecosystem functions: a global perspective. John Wiley and Sons, New York.

van Genuchten, M. Th and G. J. Hoffman. 1984. Analysis of crop salt tolerance data. In: Shainberg I and J. Shalhevet (eds.) Soil salinity under irrigation. Springer-Verlag.

Van der Valk, A. G. 1981. Succession in wetlands: a gleasonian approach. Ecology **62**:688-696.

Warner, J. H. 1990. Successional patterns in a mangrove forest in Southwestern Florida, USA. M.S. Thesis. University of Southwestern Louisiana.

Woodroffe, C. 1992. Mangrove sediments and geomorphology. In: A. I. Robertson and D. M. Alongi (eds), Coastal and Estuarine Studies. American Geophysical Union, Washington, D. C., pp. 7-41.

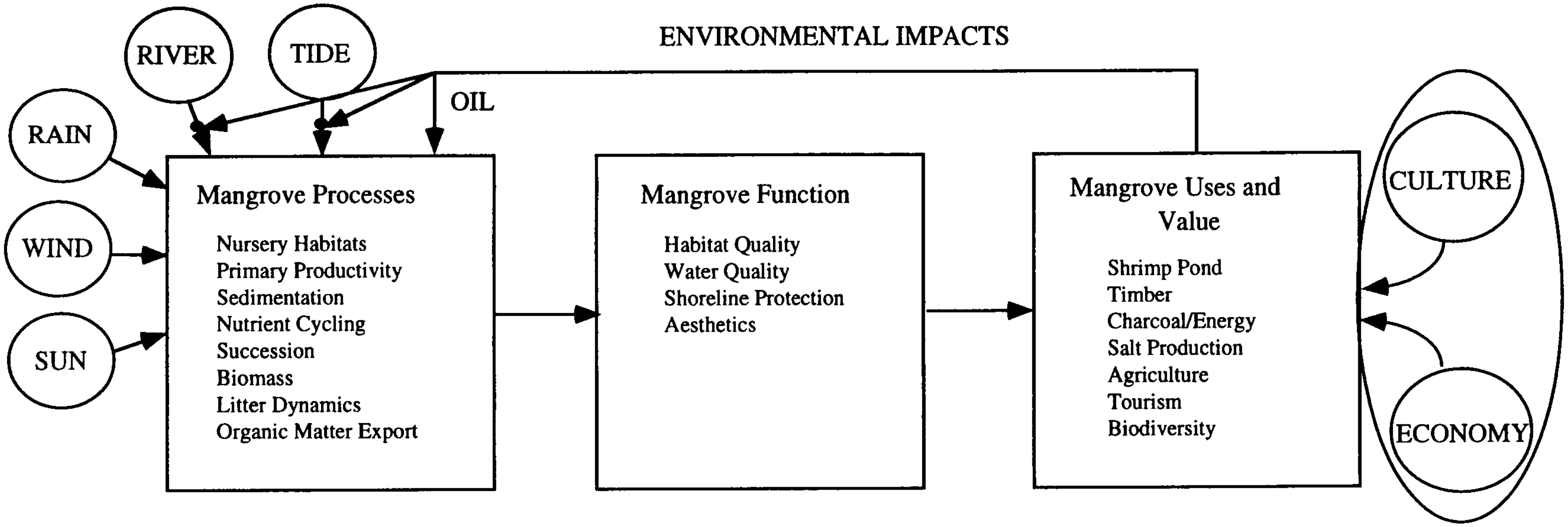


Figure 1. Linkages among the ecological processes, functions, and values of mangrove ecosystems as influenced by environmental setting and social factors including the feedback effect of oil contamination from human activities.

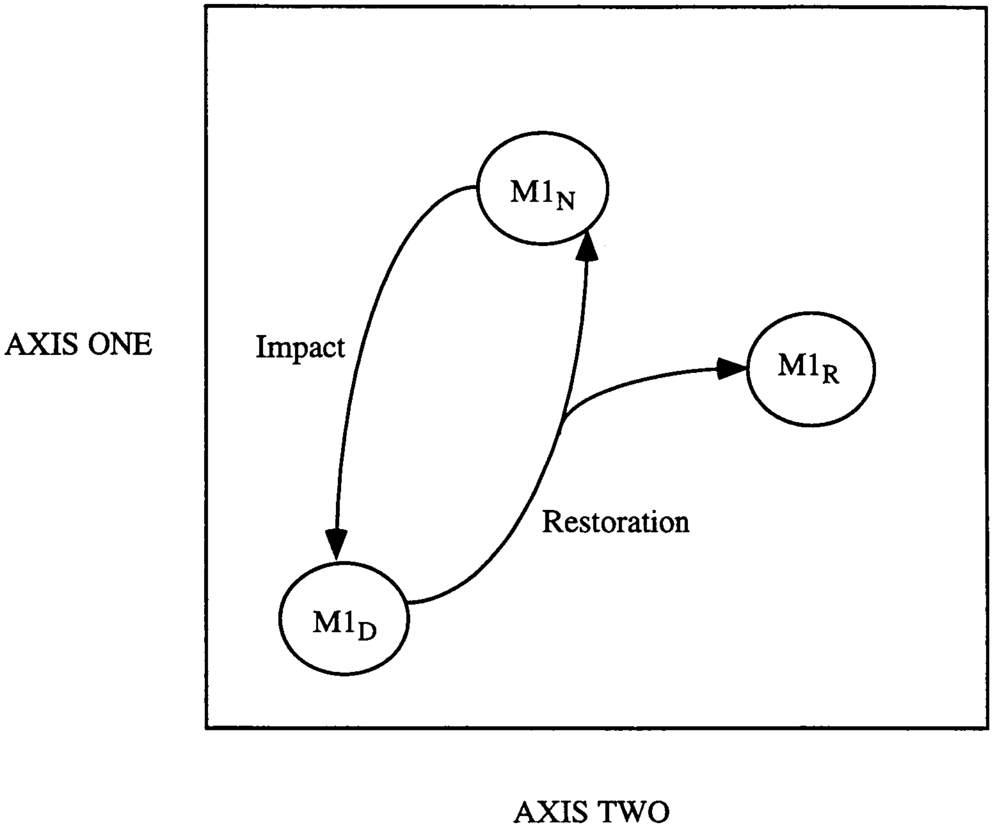


Figure 2. Trajectories of the degradation of natural mangrove wetland ($M1_N$) to a damaged condition ($M1_D$) followed recovery to either the natural state or some other new restored condition ($M1_R$).

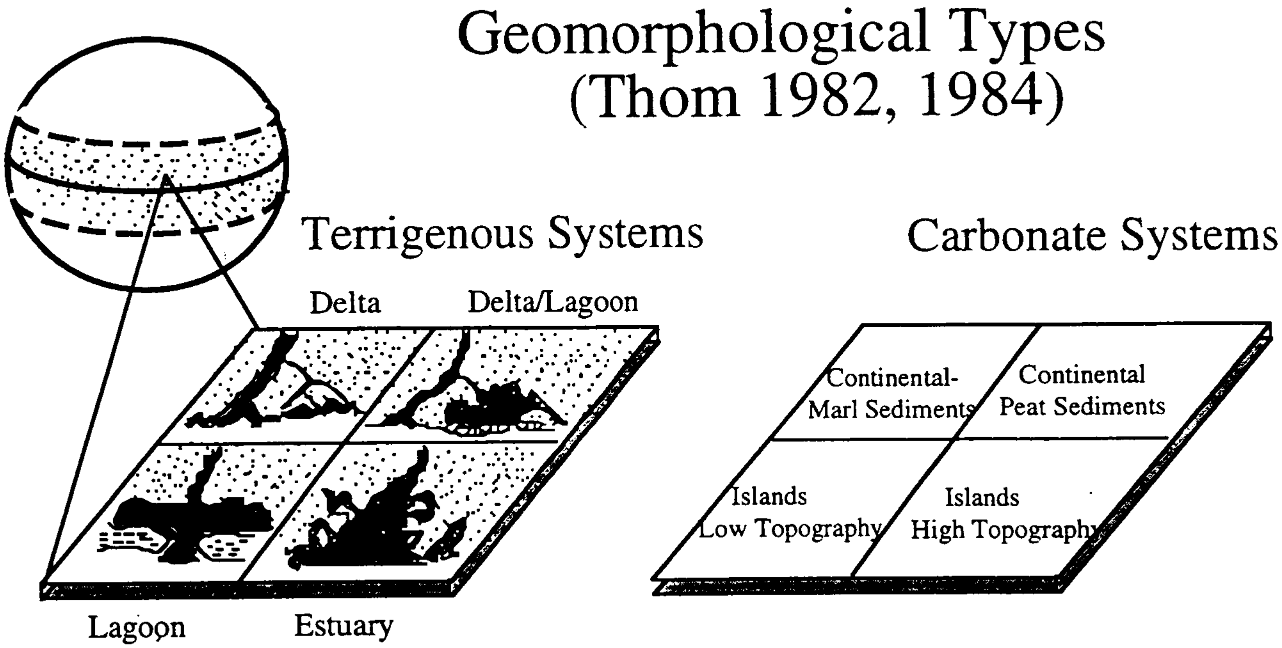


Fig. 3. Hierarchical classification system to describe properties of tropical estuaries based on global and gemorphological (regional) factors.

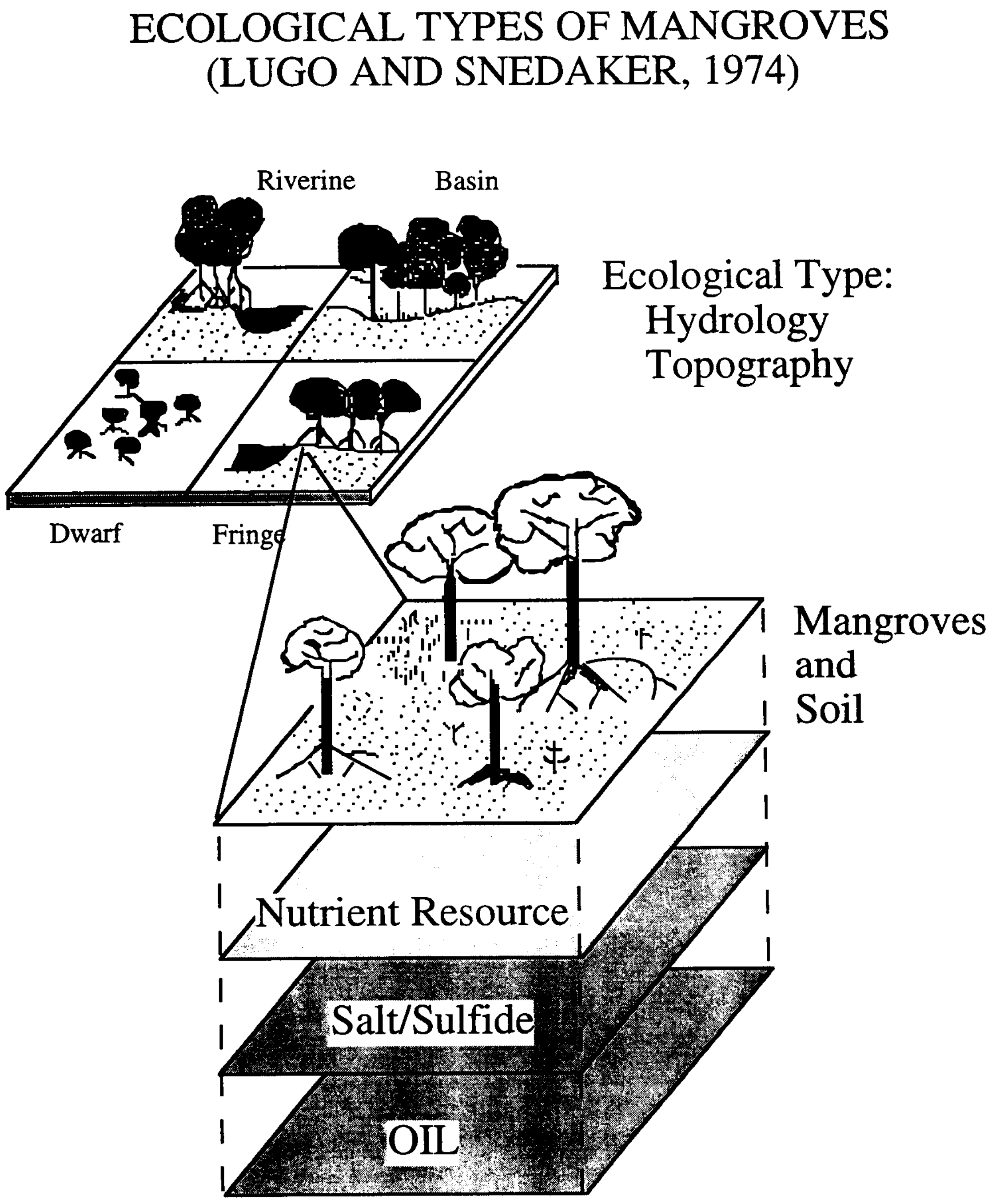


Figure 4. Hierarchical classification system to describe patterns of mangrove structure and function based on ecological (local) factors that control the concentration of nutrient resources and stressors in soil, including the distribution of oil.

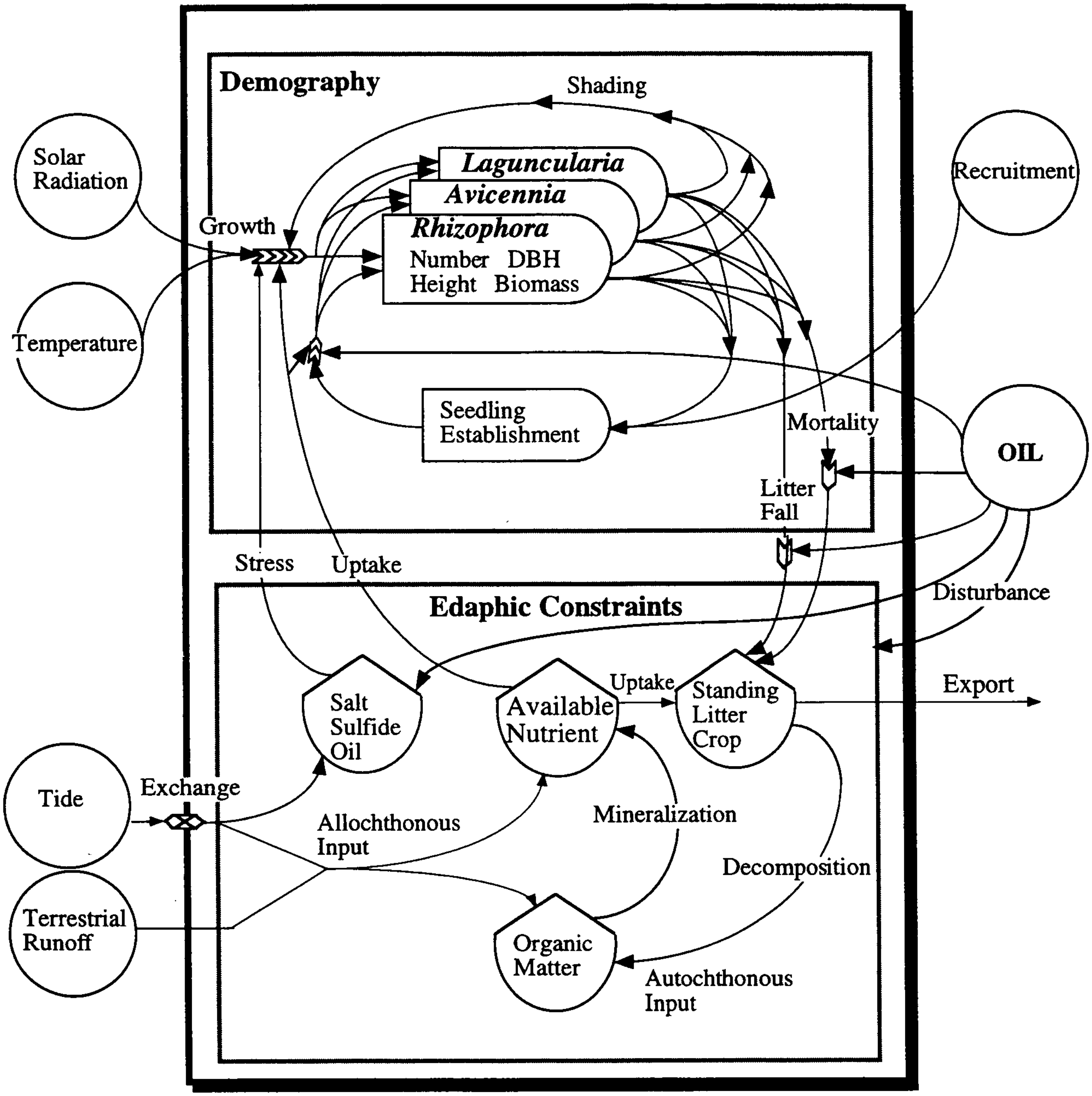


Figure 5. The conceptual model of mangrove wetland development including edaphic constrains that are influenced of oil contaminants.

MANGROVESUCCESSIONMODEL

Diagram 1. Rhizophora mangle

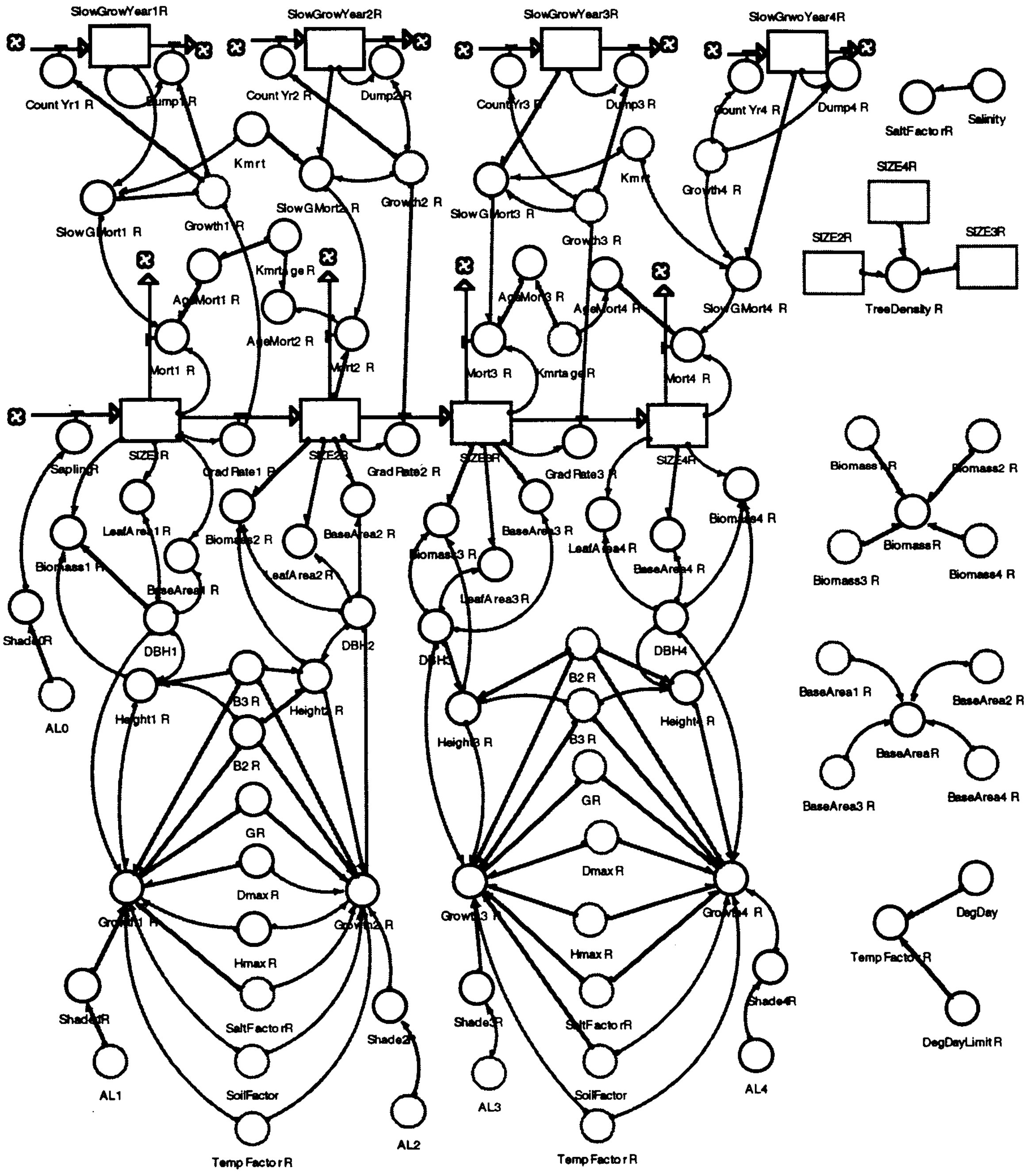


Figure 6. Stella diagram of the Rhizophora submodel of FORMAN v1.1.

Comparison output of simulation model with field data (data are expressed as basal area)

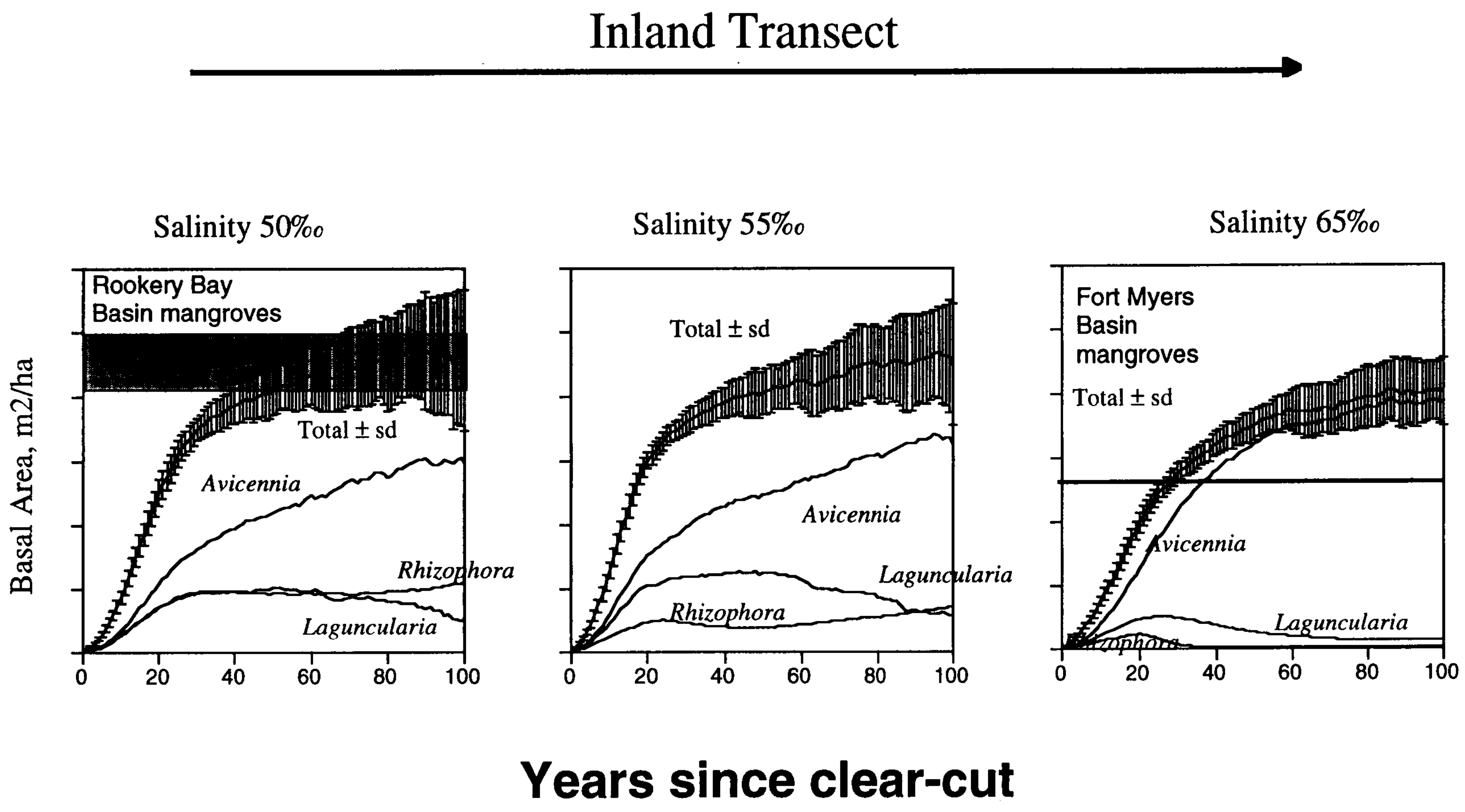


Figure 7. Comparison of simulation output with field data for Rookery Bay and Fort Myers. Lines represent field data.

Spill Containment and Cleanup

Sally C. Levings
Coastal Zone Analysis, Inc.
P.O. Box 97
Sopchoppy, FL 32308

INTRODUCTION

The mangal is a biogenic habitat -- that is the trees themselves create the habitat. Examples of biogenic habitats include mangroves, salt marshes and coral reefs. Death of the structuring organism will cause loss of the habitat, with cascading effects upon the suite of associated species. As such, mangroves, and the species dependent upon them, are vulnerable to long-term effects of oiling.

From the point of spill containment and cleanup, the fragility of the habitat is of primary concern. If oil enters a mangrove forest, cleanup needs must be balanced against the potential damage cleanup might cause to the forest itself. Our focus here is only upon mangrove forests. However, mangroves often grow in close association with other sensitive habitats (e.g. seagrass beds, coral reefs, reef flats). Decisions on cleanup and containment techniques should be made with respect to all potentially affected areas, not for single habitats in isolation.

After an oil spill, the usual first response of cleanup efforts involves attempts to contain the spilled oil, pending recovery of the spilled product. If a mangrove forest is oiled, then cleanup methods for the oiled forest must be decided upon and implemented. Here we briefly review cleanup methods used to date and suggest possible options that might be promising for reducing the effects of oiling on mangrove forests. We note that the biennial series of the Proceedings of the International Oil Spill Conferences (published by the American Petroleum Institute) contains up to date information on developments in cleanup technology and the success of different methods in field use, and are a source of extensive additional information. Despite the rapid development of cleanup technology, there are major biological, technological and resource limitations that affect removal of spilled oil from mangrove forests.

SPILL CONTAINMENT - BARRIER METHODS

Immediate mechanical exclusion and recovery of spilled oil is clearly the best way to prevent injury to the mangrove forest. 'Barrier' methods have an extensive history of use after oil spills and there is a large array of barriers that have been developed to deflect or contain oil. These range from the very simple (placement of hay bales to absorb oil before it enters an area) to the very sophisticated (open ocean booms that can deflect oil even when currents are moderately strong). In this context, burning of oil offshore is another type of "barrier" to oiling of a mangrove shoreline.

Pre-planning for placement of booms and warehousing of sufficient booms to implement plans are critical in making barrier methods work in a real-time oil spill response (refs). Oil spill response plans usually contain this information, but it is often unclear if booming will be sufficient to prevent oil entry or if booms can be deployed sufficiently rapidly to protect the shore. In addition, there are some sections of the shoreline which cannot be boomed due to their physical characteristics (i.e. current strength, water depth) and will necessarily remain vulnerable to oiling. In the confusion and chaos during a real oil spill response, even the best plans may fail due to unforeseen circumstances, rapid movement of oil, and/or human error.

While barrier methods may prevent much oiling of mangrove forests, there are both avoidable and unavoidable problems associated with booming of mangrove shorelines. Barrier methods will inevitably fail or perform inadequately in at least some circumstances. First, as we noted above, the physical characteristics of some mangrove shores prevent boom deployment. Long sections of shoreline behind shallow flats and shores with rapidly water movement are examples of locations where booms cannot usually be successfully employed. Changes in physical conditions may make booms ineffective during a spill, especially if conditions exceed those considered when planning.

Second, boom effectiveness will vary with diel and seasonal changes in water levels, winds and tides. Changes in the direction of tidal flow can trap oil behind an improperly deployed boom, forcing oil into an area. Booms that do not rise and fall with the tides can allow oil entry as water levels change.

Third, some heavy oils, especially fuel oils (e.g. #6 fuel oil), may become negatively buoyant and pass under booms (Castle et al. 1995, Michel et al. 1995, Michel and Galt 1995) while some light products (e.g. aviation fuels) mix readily into water under some conditions and may not be blocked by floating booms. Other methods of containment and cleanup must be used for these situations. Two recent examples of negatively-buoyant heavy oil spills are the 1993 Tampa Bay oil spill (Urquhart-Donnelly 1995) and the 1994 Berman spill in Puerto Rico (Burns et al. 1995).

Fourth, even when successful at keeping oil out of mangrove forests, booms may cause physical damage to the mangrove fringe (collateral damage). This is probably more likely to occur if booms are untended. If a boom line breaks or if the boom comes loose from its moorings, the boom is likely to become tangled in the mangrove fringe, breaking branches, prop roots and small juvenile and adult plants (e.g. Garrity and Levings 1994). This mechanical injury may be extensive, and at the same time, failure or entanglement of the boom potentially allows oil to enter the forest.

Finally, it may be prohibitively expensive or difficult to stockpile adequate supplies for a major spill. Pre-positioning of expensive cleanup material and trained personnel to deploy them is extremely costly and may not be realistic in many cases. If this occurs, then barrier methods are likely to prove inadequate in a real oil spill.

Shoreline Cleanup

Once oil enters a mangrove forest, cleanup methods must be decided upon. For mangroves, potential negative effects of cleanup procedures must be weighed against potential benefits of oil removal. Cleanup options are determined by the type and amount of oil, and range from no action to use of dispersants. Here we use the organization and definiations developed following the Exxon Valdez cleanup for the commonly used methods (Hayes et al.

1992). We briefly discuss how each method might be used for an oil spill in a mangrove forest, and the potential benefits and drawbacks of each.

PHYSICAL METHODS APPROVED FOR USE DURING AN OIL SPILL

1. No action

It is appropriate to do nothing when natural oil removal is rapid, when shorelines are inaccessible, or where cleanup will cause more harm than good. For mangrove shorelines, small oil spills of all oil types and those of light fuels that evaporate quickly (gasoline, aviation fuel) are candidates for no action. We note that even where no cleanup of light fuels is recommended, these products can cause significant injury requiring years to decades for recovery (Ballou and Lewis 1989). It may be best to take no action, even for a major spillage, depending upon the characteristics of the forests affected.

2. Manual removal

Manual removal is used when stranded oil (usually heavy products) can be removed with hand tools and manual labor. It is potentially appropriate in mangroves where heavy oil can be removed directly, limiting long-term sediment contamination.

Mechanical damage to prop roots, pneumatophores, trunks and branches is the major drawback of manual removal of oil in mangrove forests. In the 1993 Tampa Bay oil spill, #6 fuel oil stranded in an area of mixed mangrove forest and was removed by hand for approximately five days. Where cleanup crews manually removed oil, black mangrove pneumatophores along paths used by cleanup workers were significantly more likely to be dead than those in areas accessed by one or a few workers (cleaned, Levings and Garrity 1996). Where pneumatophores had been dense at the time of the spill, paths often were bare substrate by 15 months post-spill as broken pneumatophores died and rotted away.

Although minimizing sediment contamination will probably limit long-term effects of oiling, we know of no data that critically examines the tradeoff between physical injury from manual removal and effects of sediment oil in mangrove forests.

3. Passive collection sorbents

Sorbents are oleophilic materials that absorb oil as it floats onto or off a shoreline. Sorbents vary in their effectiveness depending upon the oil type, absorption capacity and the amount of weathering of the oil. Sorbents types vary from natural products (hay bales, treated peat-like sorbents, modified kelp) to plastics; sorbents are manufactured in forms from pellets to pom-pom booms to fabric-like sheets.

In mangroves, sorbents have been used to wipe oil from mangrove surfaces (Ibanez 1995) or in the form of booms to absorb oil released from sediments (S. Levings and S. Garrity, personal observations). With respect to these uses, considerations of physical damage from manual cleanup and boom deployment apply to use of sorbents.

There is one additional use of sorbents which may have promise for use in an oil spill in mangroves. There is a class of sorbents that are based on natural plant materials (e.g. cellulose, peat, kelp residue) that float on water and absorb oil. From the point of view of mangroves, these products might be useful at absorbing oil and preventing its penetration and deposition into mangrove sediments. Even if it proved impossible to remove these sorbents from an oiled forest, oil might be held out of sediments and be broken down relatively rapidly under aerobic surface conditions. If only natural plant products were used, the sorbent itself would decompose over time.

4. Debris removal

Manual removal of contaminated debris and drift may be used to remove sources of contamination or of bulk oil. This could be used where oil strands against a drift line of leaves and wrack and serves as a source for long-term contamination. We know of no cases where this has been used in mangroves, but the considerations listed under manual removal would also apply to removal of oiled debris.

5. Trenching

Trenching has been used to allow oil to seep from permeable substrates for collection and removal. In a mangrove forest trenching would cause direct physical injury to the forest and might serve as a conduit for further oil penetration into the forest. Trenching does not appear to be an appropriate method for use in mangroves.

6. Sediment removal

Removal of oiled sediments is used when oiled sediments are removed either manually or mechanically. Sediment removal would essentially destroy a mangrove forest and does not appear to be an appropriate method for use in mangroves.

7. Cold water flooding (Deluge)

Cold water flooding is used to wash fluid, loosely adhering oil from a shoreline for collection using low pressure water flow of ambient seawater. In a mangrove forest, it might be applicable when water can be used to flush oil out without extensive physical disturbance of sediments. However, it is not usually recommended for use in vegetated habitats (Hayes et al. 1992) and we know of no documented cases of its use in mangrove forests.

8. Cold water/Low pressure washing

Cold water/low pressure washing is used to remove liquid oil that is pooled on the surface, attached to the substrate or trapped in vegetation. Ambient seawater at low pressure is pumped to flush oil out to the water's edge for pickup using booms or skimmers.

In mangroves, low pressure washing could be used to flush oil out of the forest, if physical damage or sediment disturbance can be minimized. As recommended for marshes, flushing at water levels where sediments are submerged may help remove oil while minimizing possible physical injury to the forest. We know of no data on uses of cold water/low pressure washing in mangroves.

9. Cold or hot water/moderate to high pressure washing, Slurry sand blasting (methods 8b to 11 in Hayes et al. 1992)

These methods are not appropriate for use in mangrove forests due to their physical effects on the habitat.

10. Vacuum

Vacuum methods remove free oil pooled on the substrate or water surface; vacuum units can range from small units to large suction devices mounted on dredges. These methods are often used from outside a vegetated area.

Vacuuming free oil from outside a mangrove forest may be a technique that can remove free oil while minimizing mechanical damage to the forest. Although we know of no cases where the efficacy of vacuuming has been tested critically in mangroves, it should be considered as a possible technique to remove oil while limiting physical damage.

METHODS WHICH ARE NOT CURRENTLY APPROVED FOR USE

There are a large number of approaches which have been developed, but which have not been approved for use or for which we have no information on applicability to mangroves (see discussion in Hayes et al. 1992). However, one class of potential chemicals has tested: dispersants (including shoreline cleaners) and dispersant/oil mixtures. We note here the general concern about dispersants. Oil which is dispersed enters the water column and is put in contact with subtidal organisms that might otherwise escape exposure to oil. Given the frequent proximity of mangroves to associated habitats known to be sensitive to oil/dispersant mixtures (especially coral reefs), use of dispersant in and around mangroves must be carefully considered for the area as a whole (see for example Thorhaug et al. 1991).

1. There is limited information from field and laboratory testing of dispersants and dispersant/oil mixtures on mangroves (e.g. Getter and Ballou 1985, Getter et al. 1985, Ballou et al. 1987). In general, oil caused significantly more severe effects to mangroves than did oil/dispersant mixtures. Effects of oil alone may also persist longer than effects of oil/dispersant mixtures, since dispersed oil may be removed from sediments more rapidly than undispersed oil. However, dispersant use must be considered for the area as a whole, not just oiled mangroves, as discussed above.

2. Shoreline cleaners (Corexit 9850)

Corexit 9850, which was developed by EXXON for shoreline cleaning after the Exxon Valdez spill, has undergone limited testing for removal of viscous oil from mangroves (Teas et al. 1993). Using Bunker C, nursery plants 60-165 cm tall were coated with oil over all prop roots, trunks and major branches and cleaned at intervals ranging from 4 hours to 30 days. Corexit 9850 was applied with brushes or hand operated sprayers and rinsed with fresh seawater. Survival decreased with time the oil was left on the plant. Results were interpreted to suggest that blockage of lenticels could be tolerated by the plant for a short period, but that after approximately a week, lack of oxygen exchange would kill the plant. Difficulties with possible use of Corexit 9850 include dispersion of oil cleaned from surfaces into the water column (Hayes et al. 1992), lack of controlled experimentation on the causes of plant mortality (including possible contact toxicity of Bunker C), and lack of information on effects of Corexit 9850 on other oil types. While this approach may show some promise, issues relating to recovery of dispersed oil must be addressed before field use can be evaluated.

BIBLIOGRAPHY

Ballou, T. G., R. E. Dodge, S. C. Hess, A. H. Knap and T. D. Sleeter. 1987. Effects of a dispersed and undispersed crude oil on mangroves, seagrasses and corals. American Petroleum Institute, Publication 4460, Washington DC.

Ballou, T. G. and R. R. Lewis III. 1989. Environmental assessment and restoration recommendations for a mangrove forest affected by jet fuel. pp. 407-412 in: Proceedings of the 1989 Oil Spill Conference. API/EPA/USCG.

Burns, G. H., C. A. Benson, S. Kelly, T. Eason, B. Benggio, J. Michel and M. Ploen. 1995. Recovery of submerged oil at San Juan, Puerto Rico 1994. pp. 551-557. Proceedings of the 1995 Oil Spill Conference. API/EPA/USCG.

Castle, R. W., F. Wehrenberg, J. Bartlett and J. Nuckols. 1995. Heavy oil spills: out of sight, out of mind. pp. 565-572. Proceedings of the 1995 Oil Spill Conference. API/EPA/USCG.

Garrity, S. D. and S. C. Levings. 1994. The 10 August 1993 Tampa Bay oil spill: Injury assessment for the mangrove keys inside John's Pass. Final Report, Findings through June 1994. 140 pp. Report to NOAA Damage Assessment Center.

Getter, C. D. and T. G. Ballou. 1985. Field experiments on the effects of oil and dispersant on mangroves. pp. 577-582 in Proceedings of the 1985 Oil Spill Conference. API/EPA/USCG.

Getter, C. D., T. G. Ballou, and C. B. Koons. 1985. Effects of dispersed oil on mangroves. Synthesis of a seven-year study. Marine Pollution Bulletin 16: 318-324.

Hayes, M. O., R. Hoff, J. Michel, D. Scholz and G. Shigenaka. 1992. An introduction to coastal habitats and biological resources. Report No. HMRAD 92-4, Hazardous Materials Response and Assessment Division, NOAA.

Ibanez, M. 1995. Mangrove restoration: Cartagena, Colombia, coastal oil spill case study. pp. 990-991. Proceedings of the 1995 Oil Spill Conference. API/EPA/USCG.

Levings, S. C. and S. D. Garrity. 1996. The 10 August 1993 Tampa Bay oil spill: Injury assessment for the mangrove keys inside John's Pass. Final Report, Findings through January 1996. 193 pp. Report to NOAA Damage Assessment Center.

Michel, J. and J. A. Galt 1995. Conditions under which floating slicks can sink in marine settings. pp. 573-576. Proceedings of the 1995 Oil Spill Conference. API/EPA/USCG.

Michel, J., D. Scholz, C. B. Henry and B. L. Benggio. 1995. Group V fuel oils: source, behavior and response issues. pp. 559-564. Proceedings of the 1995 Oil Spill Conference. API/EPA/USCG.

Teas, H. J., R. R. Lessard, G. P. Canevari, C. D. Brown and R. Glenn. 1993. Saving oiled mangroves using a new non-dispersing cleaner. pp. 147-151 in: Proceedings of the 1993 Oil Spill Conference. API/EPA/USCG.

Thorhaug, A., B. Carby, R. Reese, G. Sidrak, M. Anderson, K. Aiken, W. Walker, M. Rodriquez, H. J. Teas, B. Miller, V. Gordon, F. McDonald and J. McFarlane. 1991. Dispersant use for tropical nearshore waters. pp. 415-418 in: Proceedings of the 1991 Oil Spill Conference. API/EPA/USCG.

Urquhart-Donnelley, J. 1995. The 1993 Tampa Bay spill: Preliminary assessment of natural resources. pp 24-32 in: Proceedings of the Gulf of Mexico and Caribbean Oil Spills in Coastal Ecosystems Symposium (eds. C. E. Proffitt and P. F. Roscigno). U.S. Department of the Interior, Minerals Management Service, New Orleans.

Limitations to Microbial Degradation of Hydrocarbons in Mangrove Forests

R. W. Weaver
Soils and Crop Sciences Department
Texas A&M University
College Station, TX 77843

Biodegradation of hydrocarbons requires active populations of hydrocarbon degrading microorganisms. For them to be active certain requirements must be met. A primary requirement is that degradation of hydrocarbons is an aerobic process requiring molecular oxygen (Alexander 1994). Generally it is not practical to provide other electron acceptors such as nitrate or sulfate. Other requirements for at least moderate activity are a mesophilic temperature range of 10° to 35° C and an adequate quantity of mineral nutrients. For mangrove forests low temperature would not likely pose a problem because the temperature would generally be above 20° C. It is not likely that lack of suitable microorganisms is a significant problem. The oceanic and near shore areas have received hydrocarbon pollution by either natural seeps or from spills at various times. Microorganisms are present in adequate numbers and diversity to degrade oil in the marine environment (Pritchard 1992; Rambeloarisoa et al. 1984) if conditions of nutrient supply, temperature and aeration are met. It is not likely that salinity poses a problem for microbial degradation of oil in marine environments because the responsible microorganisms are not very sensitive to salinity (Kerr and Capone 1988; Rhykerd et al. 1995).

Efforts to increase microbial activity through inoculation with commercial products containing hydrocarbon degrading microorganisms has not met with success and is not supported by scientific evidence (Pritchard 1992). Commercial cultures did not influence biodegradation of fuel oil (Dott et al. 1989) or crude oil in a sandy soil in Texas (Huesemann and Moore 1994). Addition of inoculant products to sea water in laboratory reactors or on beaches did not increase biodegradation of oil more than addition of fertilizers (Venosa et al. 1992 a,b). Application of bioremediation products did not increase populations of microorganisms degrading oil in 10 L containers containing marine sediments that were growing *Spartina* in the glasshouse (Neralla et al. 1995). The addition of oil greatly increased the populations as did addition of fertilizer to oil contaminated treatments.

The primary efforts in responding to an oil spill should be directed toward keeping the oil out of the mangrove forest and in an aerobic environment where it can degraded. Only the uppermost layer (1 mm) of sediments is oxygenated (Shelton and Hunter 1975). Absorbents may be useful in keeping the oil floating or from penetrating the sediments. The zone at the surface of the sediments is aerobic at least part of the time which makes it feasible to have reasonable high rates of degradation. In a controlled glasshouse study, using pots containing *Spartina* growing in marine sediments, intermittent flooding after allowing oil to reach the sediment surface resulted in only 40% of pre-weathered oil being biodegraded in 40 d as compared to 60% of the oil being degraded under continuously flooded conditions when oil remained floating (Wright et al. 1996a). Once oil reaches the sediments and penetrates into the sediments it is a long term problem and not likely suitable for anything but intrinsic bioremediation which is a natural process of slow degradation requiring years to accomplish

It may well be that the secondary limitation to oil biodegradation next to aeration is adequate concentrations of P and N (Atlas and Bartha 1972). Nutrient concentrations in any estuarine environment are dynamic and depend on many interactive factors (Vernberg 1993). Results of fertilization to enhance oil degradation in mangrove forests and other estuarine environments are not available. Addition of N and P to closed systems containing marine sediments increased the quantity of oil degraded in 45 d from 42% to 75%. Nutrient concentrations in the water column of containers growing *Spartina* on marine sediments was approximately 11 μM for N and less than 1.6 μM for P (Wright et al. 1996a) Fertilization with P increased oil degradation from 32% degraded in 82 d to 45% (Wright et al. 1996b). Providing Inipol ,a lipophilic fertilizer (Olivieri et al. 1978), containing both N and P increased bioremediation to 70% of the oil applied. In the summer when higher temperatures prevailed it only took 40 d for 60 % of the oil to be degraded but there was no response to fertilization (Wright et al. 1996b).

In laboratory experimentation using artificial sea water inoculated with a mixed population of oil degrading bacteria from estuarine water some insight regarding optimal P and N concentrations needed for oil biodegradation has been obtained. The results have not been published and are from R.W. Weaver and M.J. Strynar at Texas A&M University. Nitrogen concentration maintained at 300 μM was adequate for maximal rates of N uptake during oil degradation. Without constant replenishment the N became depleted to a level of approximately 10 μM in 24 h. and was not depleted further. The concentration of P that provided for maximal uptake of N during oil biodegradation was 16 μM and was depleted to near the detection limit of the methodology which was 5 μM. Values for P in seawater are approximately 2.4 μM to 1.0 μM (Gibbs 1975; Gross 1990). Concentrations of P in estuarine waters very greatly and are in the range of 14 μM to 0.29 μM (Emerson 1989; Kenosha 1988). Concentrations of N in oceanic water average 31 μM (Gross 1990) and N concentrations in estuarine waters range between in 257 μM to 43 μM (Emerson 1989; Kenosha 1988). Even though the N and P concentrations may be adequate at the time of exposure to oil the rate of replenishment may not be adequate.

LITERATURE CITED

Alexander, M.A. 1994. Biodegradation and Bioremediation. Academic Press, Inc., San Diego, California.

Atlas, R.M., and R. Bartha. 1972. Degradation and mineralization of petroleum in sea water: Limitation by nitrogen and phosphorus. Biotech. and Bioeng. 14:309-318.

Dott, W., D. Fiedieker, P. Kampfer, H. Schleibinger and S. Strechel. 1989. Comparison of autochthonous bacteria and commercially available cultures with their effectiveness in fuel oil degradation. J. Ind. Microbiol. 4:365-374.

Emerson, W.D. 1989. The nutrient status of the Sundays river estuary South Africa. Wat. Res. 23:1059-1067.

Gibbs, C.F. 1975. Quantitative studies on marine biodegradation of oil: I. Nutrient limitation at 14°C. Proc. R. Soc. Lond. B. 188:61-82.

Gross, M.G. 1990. Oceanography: A View of the Earth. 5th ed. Prentice-Hall, Inc., Englewood Cliffs, NJ.

Huesemann, M.H., and K.O. Moore. 1994. The effects of soil type, crude oil type and loading, oxygen, and commercial bacteria on crude oil bioremediation kinetics as measured by soil respirometry. *In:* R.E. Hinchee, B.C. Alleman, R.E. Hoeppel, and R.N. Miller (Eds.) Hydrocarbon Bioremediation. Lewis Publishers, Boca Raton, Florida. pp. 58-71.

Kerr, R.P. and Capone, D.G. 1988. The effect of salinity on the microbial mineralization of two polycyclic aromatic hydrocarbons in estuarine sediments. Mar. Environ. Res. 26:181-198.

Kenosha, H.M. 1988. Sources of nitrogen and phosphorus in an estuary of the Chesapeake Bay. J. Environ. Qual. 17:185-188.

Neralla, S., A.L. Wright, and R.W. Weaver. 1995. Microbial inoculants and fertilization for bioremediation of oil in wetlands. p. 31-38. *In:* R.E. Hinchee, J. Fredrickson, and B.C. Alleman (Eds.) Bioaugmentation for Site Remediation. Battelle Press, Columbus, Ohio.

Olivieri, R., A. Robertiello, and L. Degen. 1978. Enhancement of microbial degradation of oil pollutants using lipophillic fertilizers. Mar. Poll. B. 9:217-220.

Pritchard, P.H. 1992. Use of inoculation in bioremediation. Current Opinion in Biotechnology 3:232-243.

Rambeloarisoa, E., J.F. Rontani, G. Giusti, Z. Duvnjak, and J.C. Bertrand. 1984. Degradation of crude oil by a mixed population of bacteria isolated from sea-surface foams. Marine Biology 83:69-81.

Rhykerd, R.L., R.W. Weaver, and K.J. McInnes. 1995. Influence of salinity on bioremediation of oil in soil. Environ. Poll. 90:127-130.

Shelton, T.B. and J.V. Hunter. 1975. Anaerobic decomposition of oil in bottom sediments. J. Water Poll. Control Fed. 47:2256-2270.

Venosa, A.D., J.R. Haines, and D.M. Allen. 1992. Efficacy of commercial inocula in enhancing biodegradation of weathered crude oil contaminating a Prince William Sound beach. J. Ind. Microbiol. 10:1-11.

Venosa, A.D., J.R. Haines, W. Nisamaneepong, R. Govind, S. Pradhan, and B. Siddique. 1992. Efficacy of commercial products in enhancing oil biodegradation in closed laboratory reactors. J. Ind. Microbiol. 10:13-23.

Vernberg, F.J. 1993. Salt-marsh processes: A review. Environ. Toxic. Chem. 12:2167-2195.

Wright, A.L, R.W. Weaver, and J.W. Webb. 1996. Concentrations of N and P in floodwater and uptake of ^{15}N *Spartina alterniflora* in oil contaminated mesocosms. Bioresource Technology (In Press).

Wright, A.L, R.W. Weaver, and J.W. Webb. 1996. Oil bioremediation in salt marsh mesocosms as influenced by N and P fertilization, flooding, and season. Water, Air, and Soil Pollution (In Press).

Restoration of Mangrove Ecosystems After an Oil Spill

C. Edward Proffitt[1], Donna J. Devlin[2], and Thomas W. Doyle[3]

[1] Louisiana Environmental Research Center
McNeese State University
P.O. 90220
Lake Charles, LA 70609

[2] Biology Department
University of Southwestern Louisiana
Lafayette, LA 70504

[3] National Biological Service
Southern Science Center
700 Cajundome Blvd.
Lafayette, LA 70506

INTRODUCTION

Oil spills have the potential to cause severe and long-lasting disturbance in mangrove ecosystems. It is clear that once oil enters mangrove soil, often characterized by high organic content, high water levels, and low dissolved oxygen, that degradation time may be years to decades (Levings et al. 1994, Weaver et al. 1995). This produces chronic contamination that can affect survival and growth of trees and colonizing (or planted) seedlings, as well as, resulting in genetic damage to mangroves living for long periods of time in the contaminated sediment (Klekowski et al. 1994a,b and Lowenfeld and Klekowski 1992).

The appropriate management responses will be dictated by the type of oil and the extent of contamination. Here, we review possible methodologies for restoration of mangrove ecosystems following degradation by an oil spill. Remediation methods are discussed in other sections of this document and are assumed to occur, when appropriate, prior to or in conjunction with restoration.

TYPES OF RESTORATION

Local conditions and political climates have lead to the development of a variety of activities that are often lumped under the moniker of "restoration." In many cases restoration includes re-vegetation of a disturbed area or creation of "new" mangrove ecosystem which is typically coupled with creation of new intertidal habitat. In these instances, re-vegetation is often accomplished by:

(a) planting of mangroves,

(b) planting of marsh grass to serve as so-called "nurse habitat" to trap mangrove propagules, hold sediment, and provide initial wetland plants in a kind of early succession mode,

(c) allowing natural re-colonization by mangrove propagules, or in certain cases, by coppicing of surviving trees.

In other instances, "restoration" does not include any direct habitat creation or planting. Some restoration projects involve re-establishment of "normal" hydrological regimes (often for mangroves, this includes re-establishing historic tidal connections). In other cases, removal of noxious exotic plants such as brazilian pepper or australian pines have been accepted as "habitat restoration." However, since these species typically grow only at the upland (or freshwater) edge of mangrove systems, the importance of exotic removal as a form of "mangrove ecosystem restoration" is questionable.

The appropriate restoration mechanism will be case specific. In all instances, however, appropriate restoration methodologies should result in an increase in the area of mangrove forest ecosystem and/or an increase in magnitude of those ecological functions chosen as important in specific goals.

Here, we distinguish between "<u>mangrove restoration</u>" defined here as the establishment or re-establishment of a mangrove forest in order to provide ecosystem structure and function similar to that of undisturbed mangrove forests for long periods of time (decades-centuries), from "<u>mangrove silviculture</u>" which involves growing stands of mangroves for some period of time for some direct human-use benefit such as timber. Mangrove silvaculture has long been practiced in many parts of the tropics for timber, charcoal, erosion control, and other reasons (Tomlinson 1986). Consequently, workable planting, management, and harvesting techniques exist for a number of mangrove species. Mangrove restoration makes use of these, or similar, techniques.

DESIGN AND IMPLEMENTATION OF RESTORATION

1. Planning and Establishment of Restoration Goals

Restoration must be viewed in both management and scientific contexts. As an important environmental management tool, ecosystem restoration should have a series of clear and specific goals to be attained. Goals, however, are relatively worthless in science unless they are specific and measurable. For example, "re-establishment of a natural, functioning mangrove forest" is too vague and unmeasurable to be a single goal. Instead, specific goals based on established values from "natural" reference mangrove forests and the results of prior restoration projects can be developed that set the timeline by which development of desired tree size, species mix/dominance, productivity, soil development, faunal utilization, etc. can be measured.

With this in mind, the design of a restoration project should include:

(a) **Use of best known management practices** for encouraging the growth and succession of a mangrove community and

(b) **A study, based on rigorous statistical and experimental design methodologies**, to quantitatively determine if and when goals are met.

Each of these will be addressed in the ensuing sections.

2. The Mechanisms of Mangrove Restoration

a) Creation of "New" Mangrove Ecosystems

Creation of new intertidal habitat and subsequent planting or natural development of a mangrove forest on the site may be necessary when soil contamination is extensive and remediation is not expected to successfully reduce the residual oil in the soil of the disturbed site over ecologically short time frames (a few years); or, when mitigation agreements require the establishment of more forest acreage than was initially disturbed the the oil. Methods of site preparation and planting have been documented by Cintron (1992), Lewis (1982), Lewis (1990a,b), and Lugo and Cintron (1975).

Cintron (1992) reviews problems that can adversely affect success of restoration plantings. We break the list given by Cintron (1992) into "common" and "less common" groupings and combine a couple of his categories. The most common problems include:

(1) Incorrect planting elevation which is probably the single most common problem plaguing restoration planting attempts (Cintron 1992, Lewis 1982, 1990a,b, Snedaker Pers. Comm.);

(2) Exposure to excessive wave or current energy in that mangroves do not establish well unless the shoreline has a considerable degree of protection from natural fetch, boat wakes, etc.;

(3) Planting on unsuitable soil since mangroves grow best on mixtures of organics (peat), silts, and clays. Although mangroves may sometimes grow poorly on sand or be stunted on hard-packed "fill" material (Cintron 1992), this is not always the case (Proffitt and Devlin in preparation);

(4) Elevated soil salinities resulting from poor "flushing" by tides or freshwater runoff adversely affect survival and growth of mangroves (*Rhizophora mangle* and *Laguncularia racemosa* to a greater extent than *Avicennia germinans*) (Cintron 1992, McMillan 1972).

(5) Lack of nearby natural propagule source is also cited by Cintron (1992) as potentially a problem in terms of long-term recruitment into the population.

Less common, but potentially important in some locales are the following factors:

(1) Temperature extremes affect mangrove survival and growth. Excessively high temperatures can kill *R. mangle* and *A. germinans* seedlings (Cintron 1992, McMillan 1971). Similarily, very cold conditions as could occur in fairly exposed restoration sites near the high latitude limits of mangroves, may affect survival of restoration plantings;

(2) Accumulation of debris and vegetation wrack can cover and damage plantings, affect circulation, and prevent natural colonization by propagules;

(3) Disease and insect, isopod, and mammal herbivores may affect survival in some situations;

(4) Excess human intrusion and vandalism may be important in urban settings (Cintron 1992);

(5) Propagule "quality" in terms of long-term survival and growth may vary, and Cintron (1992) suggests using propagules from a nearby source forest rather than ones taken or purchased from areas further removed from the restoration site.

Creation of new mangrove ecosystems must follow a plan developed for that particular site that addresses all of the potential pitfalls listed above. Once a site location is identified that is accessable and not subject to extensive wave action, steps to take include:

(1) Survey mangrove forests in the vicinity of the planned project to determine the typical ranges of elevations above 0.0 NGVD within which the various species of mangroves are abundant and appear vigorous. Also, the slope of the natural forest should be determined. Cintron (1992) states that transverse gradients vary from 1:100 to 1:1000. The values for nearby natural sites should be used in constructing the site and in planting. Ranges of elevations for mangroves are:

    a) Red Mangrove (*Rhizohpora mangle*): 0.3-0.5 m NGVD (recommended by Lewis 1990a,b).

    b) Black Mangrove (*Avicennia germinans*): 0.43-0.60 [based on survey of volunteer seedlings by Stephen (1984)]. The upper end here might be modified for specific sites to be the highest elevation with regular tidal inundation that does not result in undue ponding of water and attendant buildup of soil salinities;

c) White Mangrove (*Laguncularia racemosa*): Relative to red and black mangroves, far fewer plantings of *L. racemosa* have been conducted. Studies of natural sites indicates that this opportunistic species has a 0.3 m NGVD mean elevation of colonizing seedlings (Detweiler et al. 1976). Proffitt and Devlin (1995) and unpublished data of Proffitt and Devlin from the site described by Stephen (1984) indicate that this species colonized (and has become numerically dominant) throughout the elevation range of mangroves on the site: 0.3-0.6 m NGVD.

(2) During this same survey, establish the slope of the natural forest (for example gradients from 1:100 or 1:1000 vertical:horizontal may often exist).

(3) Determine the tidal amplitude and period ranges from local oceanographic data of the natural forest.

(4) Compare the soils of the natural forest with the site selected for mangrove creation. Good growth of mangroves is often achieved on soils that are high in silt/clay size classes and has a relatively high organic content (Cintron 1992). Soils comprised of hard-packed dredge material or sand (Cintron 1992) or those with large amounts of rock or clay (Lewis 1990a) may not support rapid mangrove development or may even stunt surviving plants. On nutrient-poor soils, a slow release fertilizer may enhance growth rates (Cintron 1992, Darovec et al. 1975, Teas 1977).

(5) Have heavy equipment operators grade the site to the selected elevations and slope under the close supervision of the restoration manager. If the site is large, construction of one or several meandering or branching "flushing channels" may facilitate water exchange (Stephen 1984) and may enhance mangrove growth along channel edges (Proffitt and Devlin pers. field observ.). Be sure to tie final payment to proper performance to established standards. Also, check elevations and slope before the heavy equipment is allowed to leave the site (Lewis 1990a).

(6) Install plants according to goals and methodologies outlined in the site specific design of the restoration. In many cases this will involve planting one or more mangrove species. In some instances use of cordgrass (*Spartina alterniflora*) to quickly spread and stabilize sediments and to "trap" mangrove propagules may be appropriate. This latter procedure is substantially cheaper (Hoffman et al. 1982), but also creates a built-in time lag to the production of a mangrove forest. For example, photographs (Figure 9 in Lewis 1990a) of a time series following planting

showed rapid growth and apparent 100% cover by *S. alterniflora* within 2 years but incomplete cover by fairly small mangroves (most appearing in the photograph to be < 1.5 m tall) after 7 years. This can be contrasted with a site where mangroves were planted where at 7 years cover was extensive in areas with proper elevations and tree heights were typically 2.5 m and over (Proffitt and Devlin unpubl. data and in preparation). The utility of assuming relatively quick succession from marsh to mangroves as a "general rule" is also not supported by Estevez and Mosura (1985) who note that monospecific fringing stands of red mangroves in Tampa Bay killed by freezes can be replaced by marshes or remain unvegetated.

Where mangroves are to be planted, success is high with seedling propagules of *R. mangle* (Cintron 1992, Lewis 1990a). Cintron (1992) suggests collecting propagules from trees that have a well-developed abscission layer and detach easily; or, freshly-fallen healthy (intact terminal bud, no herbivore damage, not malformed, not dehydrated) propagules. In order to detect infestations by the boring beetle *Poecelops rhizophorae*, propagules can be held for several days and observed for evidence of a sawdust-like frass (K. McKee pers. comm. and D.J. Devlin, pers. observ.). Propagules should be gathered from a broad range of healthy, donor trees in order to avoid problems of reduced genetic diversity or genetically inferior stock. Propagules may be stored under moist conditions for 20+ days (Cintron 1992) and are planted to a soil depth of several centimeters. Planting seedlings is best timed to conincide with the peak of propagule availibility which is: (1) mid-August through mid-October in Florida for *R. mangle* (Lewis 1990a) or generally "late summer" throughout much of the Caribbean (Cintron 1992), (2) Summer and early Fall for *L. racemosa* in southwestern Florida (Proffitt and Devlin unpubl. data and field observations), and (3) Late Fall for *A. germinans*. Lewis (1990a) suggests planting every 2x2 m, however, 1x1 m plantings provides more rapid cover and, even though more expensive, is recommended here as the minimum spacing of seedlings when possible. Nursery grown or transplanted older plants can be used, but the cost of restoration increases considerably (Lewis 1990a). Goforth and Thomas (1980) reported that transplanted 12-18 month old seedlings were no more successful in survival and grew less over 23 months than planted propagules. Finn (1996a,b) reported successful transplantation of up to 2.5 m tall mangroves to *Bioshpere 2* with high survival rates. Digging was mostly by hand and required up to 5-6 hours for larger trees (Finn 1996a,b). Further research on survival, growth, and increased colonization through earlier reproduction of planting "older" seedling mangroves and a clear cost-benefit analysis is needed before we would recommend planting older plants in most situations.

Broadcasting seedling propagules of black (*A. germinans*) and white (*L. racemosa*) has generally not met with a great deal of success (Lewis and Haines 1981). The reason typically cited for this is a requirement for stranding without tidal flooding for periods of at least 5 (*L. racemosa*) or 7 (*A. germinans*) days (Cintron 1992, Rabinowitz 1978). Planting nursery-grown seedlings has met with success (Cintron 1992), especially when they are first acclimated to ambient salinity conditions. Where plans include planting *A. germinans* and *L. racemosa* seedlings, methods, such as first planting marsh grasses, to trap and contain mangrove propagules to be broadcasted, may speed establishment of high densities of these species.

(7) Initiate Quantitative Monitoring/Assessment Program (see later discussion). If restoration goals are not being met then perform remedial actions such as re-planting the site.

b) Re-establishment of a Mangrove Ecosystem Where One Was Eradicated by Disturbance or Pollution:

(1) Physically-disturbed mangrove forests, such as by hurricanes or bulldozers, can be restored much as described in (a) above. In these cases, site regrading of slopes and elevations will probably not be necessary, although removal of fallen trees and other debris may facilitate natural colonization.

(2) Polluted sediment, such as occurs with oil spills, is a more serious problem to same-site restoration. Oil contamination in highly organic mangrove sediments can persist, and be toxic to species in mangrove forests, for years (Duke 1991, Garrity et al. 1994, Levings et al. 1994, Snedaker et al. this report). In these cases pre-restoration studies of the extent of contamination, rate of natural breakdown of oil in the sediment, degrees of sediment toxicity to mangroves and fauna, and the effectiveness of any pollution remediation techniques applied to the site will have to be conducted in order to determine when the site is ready for ecosystem restoration. Determination of when the site is clean enough for restoration is a management decision that needs to be firmly supported by data and scientific opinions of those conducting the studies mentioned. Sediment should be free enough of contamination that it is not toxic to resident forest fauna and the mangroves. Ideally, contamination levels should be lower than those reported to increase mangrove mutation rates (see Klekowski et al. 1994a,b, Klekowski and Corredor 1995, and Lowenfeld and Klekowski 1992), although restoration should not be held up for this reason. Mangroves growing on somewhat contaminated sediment may facilitate phytoremediation of sediment possibly by, for example, pumping oxygen into the sediment. These mangroves will also retard soil loss to

erosion. However, regardless of mangrove survival and growth, a restoration project on contaminated soil should not be considered "successfully restored" until contamination levels approach background.

c) Ecological "Enhancement" of Selected Mangrove Functions in a Degraded System:

Enhancement of ecological functions usually entails removal of some physical problem such as a barrier to tidal flow or eradication of an infestation by exotic species such as Brazilian pepper and Australian pine. Such activities can produce benefits by increasing mangrove growth and productivity, and utilization by marine animals.

Exotic removal alone does not often greatly "enhance" mangrove productivity, growth, reproduction, or colonization; nor, does it affect marine faunal utilization. This is because these exotics in most cases are restricted to the edges of the mangrove system and are having their shading or crowding effects only in that area. As such, this activity, by itself, is not generally a sound form of "mangrove restoration," except in isolated instances.

Removal of barriers to tidal flow can be an important form of habitat enhancement restoration. However, in large areas, this may best be coupled with exotic removal and complete or selective replanting of the site if mangroves are stunted or if many have died in order to truly enhance the rate of ecosystem recovery.

3. Design and Implementation of Studies to Address Restoration Goals:

The "success" of restoration of a mangrove ecosystem is a function of the goals that are set and the assessment of whether or not these goals are attained. Thus, adequate design of a mangrove restoration project should include goals and metrics of these goals that extend over the entire time frame required for development of a mature mangrove forest. Estimates of this time course typically range from 20 years and up. We suggest that 20 years be viewed as a minimum time frame for goal setting and assessment, until studies of restoration and mangrove succession suggest longer (or, less likely, shorter) periods are appropriate.

In some instances requirements for success are dictated by agency permit stipulations. Often the success requirements include simply such physical factors as elevation and slope grade and biological factors such as percent survival and percent cover over some period of time (usually 3-5 years). These are certainly good variables to measure but by no means should they be used as sole criteria for success in true ecological sense. We recommend determining "success" of a restoration project through establishment of quantifiable goals in terms of environmental and biological parameters that are associated with "important" ecosystem functions. These can be established as a series of null hypotheses that can then be falsified or not falsified through studies over time that compare the restoration site with an appropriate natural reference mangal. Where possible, we further recommend that studies include one or more carefully selected natural mangrove systems as "reference sites." The reference site is used as a yardstick by which to judge the rates of attainment of values typical of mangrove systems for a suite of ecological variables.

Many biological and physiochemical features of mangrove systems can be selected for studies that can provide important data on restoration success. Below, we provide a recommended list, but recongnize that each restoration project study need not contain all of these. Our list of variables to include is not exhaustive but reflects our experience and study preferences.

A. Initial Data (after site preparation, but before planting): A mapping of the spatial dispersion of these variables over the study site may help explain localized differences in survival and growth (and other variables) later.

1. Site elevations and slope
2. Sediment grain size, organic content, and nutrient concentrations.
3. Tidal inundation (depth and frequency).

B. Post Planting (or post "natural colonization" for unplanted sites) Data:

1. Physiochemical factors to be quantified
    - a) Annual (if necessary) analyses of sediment grain size and especially of organic content.
    - b) Salinity of standing and pore water.
    - c) Air and sediment temperatures in open areas and below plants.
    - d) Percent light penetration to ground level.
    - e) Tidal inundation (depth and frequency).
    - f) Rainfall and other freshwater inputs.
    - g) Nutrients

2. Biological factors
    - a) Survival of planted mangroves and those colonizing the site. Tagging should be used to ensure that these can be distinguished.
    - b) Growth (at least height and trunk diameter) and reproductive output of mangroves.
    - c) Density of planted and volunteer plants. Also, dominance in the canopy and leaf area index should be determined.
    - d) Litterfall and litter "fate" (% exported vs % decomposing in situ) should be assessed. Primary mechanisms of decomposition including relative importance of microbes and invertebrate littervores should be studied.
    - e) Colonization and population densities of selected "typical" mangrove fauna (to include species in the canopy, on the forest floor, and those encrusting prop roots) should be studied.
    - f) Population sizes and grazing rates of important herbivores especially insects and the crab *Aratus pisonii.*

LITERATURE CITED

Cintron, G. 1992. Restoring mangrove systems. pp. 223-277. In, G. Thayer (ed.), Restoring the Nation's Marine Environment. Maryland Sea Grant College Publ. College Park, MD.

Darovec. J.E., J.M. Carlton, T.R. Pulver, M.D. Moffler, G.B. Smith, W.K. Whitfield, C.A. Willis, K.A. Steidinger, and E.A. Joyce. 1975. Techniques for coastal restoration and fishery enhancement in Florida. Fla. Dep. Natur. Resour. Mar. Res. Publ. 15.

Detweiler, T.E., F.M. Dunstan, R.R. Lewis, and W.K. Fehring. 1976. Patterns of secondary succession in a mangrove community, Tampa Bay, Florida, pp. 52-81. In: R.R. Lewis (ed). Proc. Second Annual Conf. Restoration of Coastal Vegetation in Florida. Hillsborough Community College, Tampa, FL.

Duke, N. 1991. Mangrove Forests. Pages 153-178 *in* B. Keller and J.B.C. Jackson, editors. Long-term assessment of the oil spill at Bahia Las Minas, Panama. Interim Report. Vol. II: Technical Report. U.S. Dept. of the Interior, Minerals Mgmt. Serv. OCS Study MMS 90-0031. 450 pp.

Estevez, E.D. and L. Mosura. 1985. Emergent vegetation. pp. 248-278. In, S.A. Treat, J.L. Simon, R.R. Lewis, and R.L. Whitman (eds.). Proceedings Tampa Bay Area Scientific Information Symposium. Sea Grant Project No. IR/82-2. Bellwether Press.

Finn, M. 1996a. The mangrove mesocosm of Bioshpere 2: Design, establishment, and preliminary results. Ecol. Engineering 6:21-56.

Finn, M. 1996b. Comparison of mangrove forest structure and function in a mesocosm and Florida. Ph.D. Dissertation, Georgetown University, Washington, D.C. 309 pp.

Garrity, S.D., S.C. Levings, and K.A. Burns. 1994. The Galeta oil spill I. Long-term effects of the physical structure of the mangrove fringe. Estuarine Coastal and Shelf Science **38**:327-348.

Goforth, H.W. and J.R. Thomas. 1980. Plantings of red mangroves (*Rhizopohra mangle* L.) for stabilization of marl shorelines in the Florida Keys. pp. 207-230. In, D.P. Cole (ed.). Proceedings 6th Ann. Conf. Wetlands Restor. Creation. Hillsborough Community College, Tampa, FL.

Hoffman, W.E., M.J. Durako, and R.R. Lewis. 1982. Habitat restoration in Tampa Bay, pp. 161-657. In: S.A. Treat, J.L. Simon, R.R. Lewis, and R.L. Whitman (eds.). Proceedings Tampa Bay Area Scientific Information Symposium. Sea Grant Project No. IR/82-2. Bellwether Press.

Klekowski, E.J. and J.E. Corredor. 1995. Is genetic degradation of mangroves a consequence of petroleum? pp. 99-105. In: C.E. Proffitt and P.F. Roscigno (eds.). Proceedings - Gulf of Mexico and Caribbean Oil Spills in Coastal Ecosystems: Assessing Effects, Natural Recovery, and Progress in Remediation Research. OCS Study/MMS 95-0063. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA.

Klekowski, Jr., E.J., J.E. Corredor, J.M. Morell and C.A. Del Castillo. 1994a. Petroleum pollution and mutation in mangroves. Mar. Pollut. Bull. **28**:166-169.

Klekowski, Jr., E.J., J.E. Corredor, R. Lowenfeld, E.H. Klekowski and J.M. Morell. 1994b. Using mangroves to screen for mutagens in tropical marine environments. Mar. Pollut. Bull. **28**:346-350.

Levings, S.C., S.D. Garrity, and K.A. Burns. 1994. The Galeta oil spill. III. Chronic reoiling, long-term toxicity of hydrocarbon residues and effects on epibiota in the mangrove fringe. Estuarine Coastal and Shelf Science **38**: 365-395.

Lewis, R.R. 1982. Mangrove forests, pp. 153-171. In, R.R. Lewis (ed.), Creation and Restoration of Coastal Plant Communities. CRC Press, Boca Raton, FL.

Lewis, R.R. 1990a. Creation and restoration of coastal plain wetlands in Florida. pp. 73-101. In: J.A. Kusler and M.E. Kentula (eds.), Wetland Creation and Restoration. EPA/600/3-89/038. Island Press, Washington, D.C.

Lewis, R.R. 1990b. Creation and restoration of coastal wetlands in Puerto Rico and the U.S. Virgin Islands, pp. 103-123. In: J.A. Kusler and M.E. Kentula (eds.), Wetland Creation and Restoration. Island Press, Washington, D.C.

Lewis, R.R. and K.C. Haines. 1981. Large scale mangrove restoration on St. Croix, U.S. Virgin Islands-II. Second year, pp. 137-148. In: D.P. Cole (ed.), Proceedings of the Seventh conference on restoration of coastal vegetation in Florida. Hillsborough Community College, Tampa, FL.

Lowenfeld, R. and E.J. Klekowski. 1992. Mangrove genetics. I. Mating systems and mutation rates in *Rhizophora mangle* in Florida and San Salvador Island, Bahamas. Int. J. Plant Sci. **13**:394-399.

Lugo, A.E. and G. Cintron. 1975. The mangrove forests of Puerto Rico and their management, pp. 825-846. In: G.E. Walsh, S.C. Snedaker, and H.J. Teas (eds.), Proceedings of an International Symposium on the Biology and Management of Mangroves. Inst. Food Agric. Sci., Univ. of Florida, Gainesville, FL.

McMillan, C. 1971. Environmental factors affecting seedling establishment of the black mangrove on the central Texas coast. Ecology **52**:927-930.

Proffitt, C.E. and D.J. Devlin. 1995. Survival, growth, and succession in a southwestern Florida mangrove restoration site: Years 6-13. (abstract). pp. 105-106. In: Estuaries: Bridges from Watersheds to Coastal Seas. Abstracts of 1995 Estuarine Research Federation meetings.

Rabinowitz, D. 1978. Dispersal properties of mangrove propagules. Biotropica **10**:47-57.

Stephen, M.F. 1984. Mangrove restoration in Naples, Florida, pp. 201-216. In: F.J. Webb (ed.). Proc. Tenth Ann. Conf. Wetl. Restoration and Creation. Hillsborough Community College, Tampa, Fl.

Teas, H.J. 1977. Ecology and restoration of mangrove shorelines in Florida. Environ. Conserv. **4**:51-58.

Tomlinson, B. 1986. The Botany of Mangroves. Cambridge University Press, Cambridge. 413 pp.

Weaver, R.W., B. Crites, S. Neralla, A. Wright, and J.W. Webb. 1995. Evaluation of commercial bioremediation products for oil biodegradation in salt marshes, pp. 165-173. In: C.E. Proffitt and P.F. Roscigno (eds.). Proceedings - Gulf of Mexico and Caribbean Oil Spills in Coastal Ecosystems: Assessing Effects, Natural Recovery, and Progress in Remediation Research. OCS Study/MMS 95-0063. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA.

The Department of the Interior Mission

As the Nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural resources. This includes fostering sound use of our land and water resources; protecting our fish, wildlife, and biological diversity; preserving the environmental and cultural values of our national parks and historical places; and providing for the enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to ensure that their development is in the best interests of all our people by encouraging stewardship and citizen participation in their care. The Department also has a major responsibility for American Indian reservation communities and for people who live in island territories under U.S. administration.

The Minerals Management Service Mission

As a bureau of the Department of the Interior, the Minerals Management Service's (MMS) primary responsibilities are to manage the mineral resources located on the Nation's Outer Continental Shelf (OCS), collect revenue from the Federal OCS and onshore Federal and Indian lands, and distribute those revenues.

Moreover, in working to meet its responsibilities, the **Offshore Minerals Management Program** administers the OCS competitive leasing program and oversees the safe and environmentally sound exploration and production of our Nation's offshore natural gas, oil and other mineral resources. The MMS **Royalty Management Program** meets its responsibilities by ensuring the efficient, timely and accurate collection and disbursement of revenue from mineral leasing and production due to Indian tribes and allottees, States and the U.S. Treasury.

The MMS strives to fulfill its responsibilities through the general guiding principles of: (1) being responsive to the public's concerns and interests by maintaining a dialogue with all potentially affected parties and (2) carrying out its programs with an emphasis on working to enhance the quality of life for all Americans by lending MMS assistance and expertise to economic development and environmental protection.