PERSPECTIVES ON THE MARINE ENVIRONMENT

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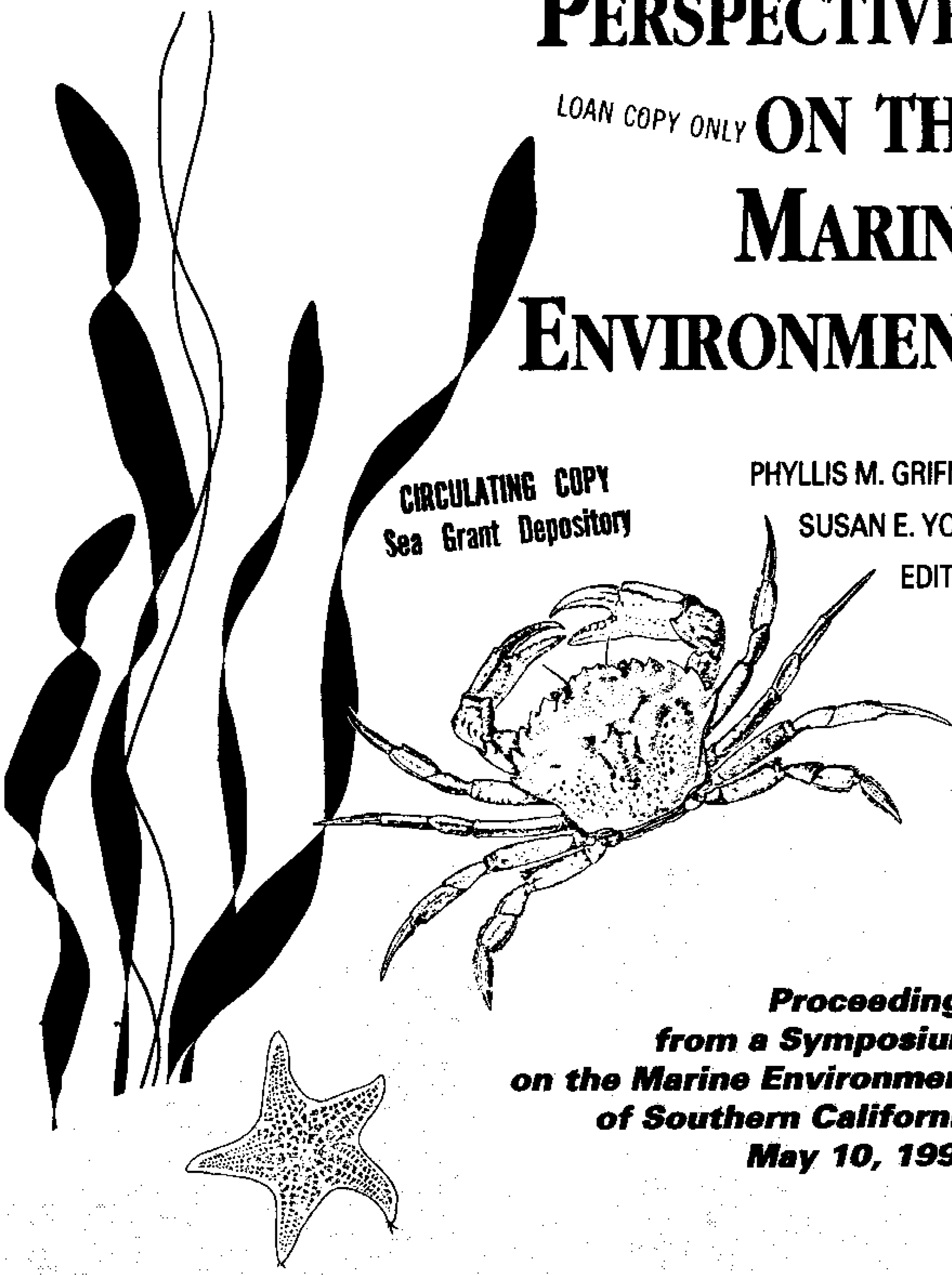

Proceedings from a Symposium on the Marine Environment of Southern California May 10, 1991

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Phyllis M. Grifman and Susan E. Yoder

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Los Angeles, California

100th Anniversary Meeting of the Southern California Academy of Sciences

Sea Grant Program
University of Southern California

USCSG-TR-01-92

Published by the Sea Grant Institutional Program, Hancock Institute for Marine Studies, University of Southern California, Los Angeles, California, June 1992. Copies are available from: USC Sea Grant Program, University Park, Los Angeles, CA 90089-1231. (213) 740-1961, FAX (213) 740-5936.

This publication has been produced with support from the National Sea Grant College Program, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, under grant number NA 90 AA-D-SG525, and by the California State Resources Agency. The views expressed herein do not necessarily reflect the views of NOAA or any of its sub-agencies. The U.S. Government is authorized to reproduce and distribute copies for governmental purposes.

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Introduction

The collection of papers in this volume, "Perspectives on the Marine Environment," was originally presented at the 100th Anniversary Meeting of the Southern California Academy of Sciences, May 10-11, 1991, in Los Angeles, California. The special symposium on "The Marine Environment of Southern California" was sponsored by the University of Southern California Sea Grant Program.

Sea Grant is a marine research and outreach program funded by the United States Department of Commerce through the National Oceanic and Atmospheric Administration. It operates in 29 major research universities around the nation, including the University of Southern California. In addition to funding research, it also provides outreach through Marine Advisory Services and Communications programs. Sea Grant sponsors research in biology, geology, oceanography, engineering, economics, urban planning, marine policy, and political science, often in an interdisciplinary context.

The objective of this volume is to develop an understanding of the range of problems associated with monitoring and managing the southern California coastal marine environment, and to suggest ways in which we can improve existing efforts. The subject matter is broad in scope, encompassing local contamination trends, health of fish populations in a local marina, problems posed by invasions of exotic plants in coastal wetlands, methods of enhancing coastal fisheries by means of harvest refugia and man-made reefs, ways to improve monitoring methods of marine biota, and problems associated with major oil spill clean up efforts. Authors were requested to apply scientific evidence in their fields of expertise toward developing better management strategies, and we think they were eminently successful. The information presented here should be useful not only to scientists in these specialities, but also to managers, industries, public regulators, and environmental groups.

We begin with a paper by Alan Mearns which provides us with an historical perspective of chemical contamination in the Southern California Bight. Based on a study conducted by the Office of Ocean Resources of the National Oceanographic and Atmospheric Administration, the author suggests which contaminants are of concern, which are not, and which require more study to determine their potential impacts. The results are in some cases surprising and contrary to public perception of the problem. Contaminants in bays and harbors are far higher than those of ocean outfalls and open coastal systems. The author suggests that priorities for the surveillance and management of our coastal marine environment should be shifted to the more polluted areas of our coast.

This paper is followed by a study of fish populations in Marina del Rey, completed by Dorothy Soule of the Hancock Institute for Marine Studies at USC and the Vantuna Research Group at Occidental College led by John Stephens. The authors suggest that harbors and marinas, despite contamination problems, may still support diverse fish communities even when sensitive invertebrate species have been diminished. The authors also suggest design improvements for marinas to improve their function as a marine nursery and fish habitat.

In the third paper of this volume, Joy Zedler of the Pacific Marine Estuarine Laboratory of San Diego State University addresses the threat posed to coastal wetlands by invasions of exotic plant species. Wetlands preservation is a particularly sensitive issue in California and the nation, given the high value placed on coastal wetlands as marine nurseries for fish and invertebrates and wintering habitat for migratory birds, and the high losses of native wetlands sustained in this century. The author describes the major causes behind the spread of particular exotic species into this community, and makes recommendations for preventing the problem.

The next two papers address methods of managing coastal fisheries. In the first, Mark Carr and Daniel Reed of the University of California at Santa Barbara address an important issue in managing reef fish populations in southern California — enhancing recruitment in nearshore fisheries. The authors describe four patterns of recruitment of reef fish in the Southern California Bight, and suggest design criteria for each pattern. An important implication is that no single refuge will be likely to improve recruitment of all southern California reef species.

The next paper, by Tom Johnson of the Port of Long Beach and colleagues Barnett, DeMartini and Purcell, investigates the ecologic value to fish communities of Torrey Pines Artificial Reef in San Diego County. Their results suggest that artificial reefs can provide valuable living space, nursery habitat, and food resources to local reef species. Far from being only a fish attractor, Torrey Pines Artificial Reef provides food for target species two orders of magnitude higher than in sandy habitats. Both somatic and gonadal production of Torrey Pines reef fish communities are also substantially higher than in sandy habitats.

The following two papers suggest ways to improve monitoring of marine environments. In the first, Craig Osenberg, Sally Holbrook and Russell Schmitt of the Marine Science Institute of the University of California at Santa Barbara suggest ways to improve the rigor of impact assessments. Impact assessments are widely used to ascertain effects of localized anthropogenic discharges. The authors describe three sampling designs now in use for assessing marine biota, and the potential for interpretational error in each design. They suggest that impact assessments be based on a new design: the Before-After-Control-Impact-Paired (BACIP) assessment design. The authors also suggest methods to improve the statistical power of this design.

The next paper, by Pat Baird of the Kahiltna Research Group at California State University, Long Beach, assesses the use of seabirds as bio-indicators of the status of the health of the ocean. The author outlines the problems associated with monitoring marine species in general due to the nature of the oceanic environment, and suggests ways in which high trophic-level predators such as seabirds can be of use in monitoring the status of both biotic and abiotic factors in these environments.

The final paper in this volume, submitted by Dennis Lees of Ogden Environmental and Energy Services, addresses a controversial topic: how an oil spill cleanup should be conducted. The massive EXXON VALDEZ oil spill is presented at length as a case study in which most opportunities for studying treatment alternatives were lost due to "poor planning before the spill; poor implementation of good planning; the confusion surrounding the spill; the inexperience, ill will, or divergent motives of many of the participants; and...the litigative environment." In his conclusions he suggests some guidelines for improving our future handling of oil spills.

We wish to thank the Southern California Academy of Sciences, especially Dr. Camm Swift, for assistance in organizing this symposium. We also wish to thank Dr. Alan Mearns, author of one of the papers in this volume, who co-chaired the symposium and was instrumental in putting it together. Finally we wish to thank all those who participated in this symposium and helped to make it a success.

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Contaminant Trends In The Southern California Bight: Four Decades Of Stress And Recovery

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Abstract. Existing and historical data were identified, collected, reviewed and reassembled to develop a region-wide, long-term history of contaminatinn of fish, shellfish and sediments of the bight and adjacent areas of Mexico. Spatial and temporal trends were developed for 10 metals (Ag, As, Cd, Cu, Hg, Pb, Se, Sn, Zn), PCBs, organotins, aromatic hydrocarbons and pesticides including DDT, chlordanes and dieldrin. Dated cores confirm that most contaminants increased during the 1950's and '60's and decreased during the 1970's and 1980's. Despite past major inputs followed by vigorous source control, metals in fish have not been elevated and have not changed. However, high levels of pesticides have been reduced nearly 100-fold. Curiously, concentrations of several metals in mussels have been higher at offshore islands than near "polluted" urban areas. Although lead in mussels has declined in all areas, concentrations remain abnormally high in the Los Angeles area.

Environmental inputs and levels of most pollutants in the open coastal zone are now declining to what they were 30 to 40 years ago. Sediments and marine organisms of bays and harbors are considerably more contaminated than those from the open coastal zone and generally more contaminated than at coastal outfall areas. Systematic surveillance should be continued to make sure concentrations in the open coastal zone remain reasonably low and that new kinds of contaminants aren't increasing. Likewise, surveillance and management of bays and harbors should be greatly increased. Finally, there is urgent need for a centralized regional data center that is responsible for acquiring and archiving existing and new information. A high rate of retirement and the death of performing scientists creates the urgency.

INTRODUCTION

This paper reviews findings from a recently completed project that reconstructed long-term trends and region-wide patterns of chemical contamination in sediments and marine life of the Southern California Bight [1]. Here I attempt to condense the findings of that report by addressing some of the fundamental questions about the quality of our coastal zone: Is contamination increasing, decreasing or remaining unchanged? How contaminated are fish, shellfish and other marine resources now and where are the most contaminated areas today? What contaminants are of most and least concern? And, finally, what changes are indicated for surveillance and management.

Each year public agencies and industry spend hundreds of millions of dollars to reduce coastal pollution and to conform with a growing array of laws and regulations. In addition, several tens of millions are spent on monitoring programs designed to measure the benefits of various controls and to substantiate the need for additional legislation and enforcement [2]. Rarely, however, does the public become aware of what these programs are finding because the information is not pulled together and assessed, nor the results published on a continuing basis. Instead, we are treated to occasional news items about this or that "hot spot." Rarely are individual pollution stories put into the larger context of pollution control successes and failures. Without these larger assessments, the public and their representatives are not aware of the progress of pollution management programs and where additional efforts are needed. This situation can result in ineffective policies where funds are expended on areas that require no further control while more critical problems remain under-studied and under funded.

Nowhere is this more apparent than in our misunderstanding of chemical contamination of marine resources along the southern California coast. It is generally assumed by the public that contamination of seafood and other marine life is extensive and getting worse and that all the materials we discharge to the ocean are in fact contaminating marine organisms. In contrast, monitoring data show just the opposite, at least for the open coastal zone: where waters, sediments and marine life were once highly contaminated ten to twenty years ago, levels of contamination have since declined. And, materials such as some so-called "heavy metals" we once thought were accumulated to high concentrations through marine food chains are not accumulated at all in marine ecosystems of the bight. However, there is much less certainty about trends in bays and harbors.

These are just some of the findings of a recently-completed assessment of contaminant trends in the Southern California Bight conducted by the Office of Ocean Resources and Conservation Assessments (ORCA) of the National Oceanic and Atmospheric Administration (NOAA [1]). These findings should be viewed as extremely important to the public and their representatives since (1) they are measures of the successes and failures of specific pollution abatement programs and (2) indicate where problems are resolved and where they are not. Below I attempt to summarize some of the more salient features of our assessment and also offer some of my own conclusions.

APPROACH

The NOAA assessment focused not only on collecting and reviewing existing contaminant monitoring data but also on ways to assemble the many pieces into a more coherent history of contamination including a simple but direct method for evaluating current contaminants of concern.

To complete this assessment, it was necessary to conduct a region-wide data search, acquire appropriate data sets, organize them into readily accessible files, reassemble selected records into comparable units of time, geography, species, and tissue types, and then make comparisons among geographic regions and among various points in time. These steps are outlined below and in more detail in [1].

Contaminants. Contaminants selected for review included 10 trace elements, a suite of polycyclic aromatic hydrocarbons (PAHs), the polychlorinated biphenyls (PCBs), and 3 historically important pesticide groups (DDT, chlordane and dieldrin). In addition, a brief scan of selected data sets was done to identify all chemicals searched for and reported in regional and local monitoring programs. These included several dozen volatile organics (such as chemicals found in cleaning solvents) and several groups of extractable organics such as phenols.

Geographic Setting. The study was limited to the Southern California Bight which includes that portion of the pacific coast of North America extending from Point Conception, California, south to Cabo Colnett, Baja California, Mexico and seaward to the break of the mainland shelf. The waters of the bight overlay a submerged continental borderland interlaced with a series of deep silty basins and surfacing mountain ranges that form offshore islands radiating over 100 km seaward from the Los Angeles area. However, the salient oceanographic feature of the bight that distinguishes it from all coastal areas to the north and south is the eastward indentation of the coastline that allows for a northward flowing return eddy, the counterclockwise Southern California Gyre. As a result of this gyre, the bight is an enclave of regionally specific populations of marine life, a trap for warm water, and a reservoir for materials entering from the land, air, and sea. It is chiefly this "stall" in the current systems of the Pacific, coupled with an adjacent urban population of 15 million, that creates cause for concern about contaminant loading and pollution effects on a region-wide basis.

The combined U.S.-Mexico population intentionally and inadvertently discharges much of the region's wastewaters to the coastal zone of the bight. Although total wastewater emissions have been increasing, inputs of many potential pollutants have been decreasing over the past two decades, a direct result of increased source control and treatment [3]. However, many other sources of pollutants (harbors, urban runoff, aerial fallout) remain poorly studied and of increasing concern.

For this study, data on contaminants in sediments and marine organisms were sought for all bays, lagoons, harbors, and estuaries as well as from basins, islands, and the coastal shelf of southern California and northern Baja California.

Types of Data. The study was limited to sediments, invertebrates and fish. Sediment data searches included information from dated sediment cores from which environmental trends could be deduced. For invertebrates, special attention was given to data from numerous "mussel watch" surveys. However, additional data was collected on contaminants in many other invertebrates including edible species of clams, crabs, shrimp and echinoderms. Data were also collected for many species of pelagic and nearshore fishes, sharks and rays. Data were not otherwise limited by tissue type, collecting methods, or chemical extraction and analytical techniques.

Data Compilation. Original ungrouped concentration values for individual samples or composites were sought for each report. In addition to the contaminant values, data and auxiliary information on sampling and analytical procedures were sought and actively collected in any available form including published and unpublished reports, memos, laboratory records, raw data sheets, and magnetic tapes and diskettes. Data on contaminant concentrations, sample and sampling characteristics, and analytical methods were extracted into a common format and then entered into desktop computer data base management systems specifically developed for this purpose. Means medians, ranges, and standard deviations of contaminant concentrations in sediments mussels, other macroinvertebrates (shellfish), and fish were computed and listed for individual surveys, geographic regions, time periods, species, and tissues. Geographic differences among bays and other survey units were further revealed by tabulating and comparing median values and noting apparent differences.

Assessment of Contaminants of Concern. Special effort was devoted to making use o this large amount of data to evaluate which contaminants were of most and leas concern. A "preponderance of evidence" approach was used. The primary challenge was that although each of the contaminants selected for review is a hazard (potentia threat) under specific conditions, those conditions may or may not be met in the bight Contaminants of most concern in sediments and organisms of the bight should have al or most of the following properties: 1) accumulate to toxic levels in sediments, 2 accumulate in excess in plants, invertebrates or fish either to the detriment of the fish or at concentrations that pose risk to seafood consumers, 3) biomagnify through marine food webs and 4) have inputs that are increasing or expected to increase. The corollary is that contaminants of least concern are those that do few or none of these with emphasis on lack of demonstrable bioaccumulation and toxicity receiving considerable weight. For example, while some materials may be undergoing increased inputs they may be of little concern because they are non-conservative, never accumulate in excess in water, sediments or biota. Likewise, a contaminant is of low concern if although it is elevated in sediments, it is neither contributing to sediment toxicity no is it accumulating in marine organisms. To qualify for this assessment, a contaminan must be in excess, which implies that there are adequate reference or control value available.

THE DATA BASE

At least 150 local, state, and federal survey and monitoring activities have been conducted to measure contaminants in sediments, invertebrates, and fish since the earl 1960s. All the surveys used in the assessment are described in [1]. Long-term trend were largely derived from sediment core profile data (such as [4-7], wastewater distric compliance monitoring programs [8,9] and the California Mussel Watch [10]. Large scale (region-wide) and local contaminant distribution patterns were derived fron numerous sources. Examples of synoptic sediment surveys include the 1977 and 198 "60-meter" and "Reference" surveys [11,12], San Pedro Bay and Basin Surveys [13 and the NOAA National Status and Trends Program [14]. Examples of "mussel watch surveys include the National Pesticide Monitoring Program [15], 1971 and 1974 coasta mussel surveys [16], mussel surveys in Baja California (as summarized in [17]) and the California Mussel Watch [10]. Examples of fish and other macro-invertebrate survey used include regional dover sole (*Microstomus pacificus*) surveys [18], surveys of con taminants in seafood organisms [19] and the NOAA NS&T Benthic Surveillance Program [20]. There are many more as described in [1]. The number of sediment samples is estimated to be in the range of 6,000 to 8,000. Samples of marine life for tissue analysis are in the range of 6,000 to 8,000. For organic chemicals alone the estimate is about 5,000, of which 4,500 have been extracted and entered into NOAA's data base systems.

LONG-TERM TRENDS: IS THE COAST CLEANER?

With one interesting exception, the long-term trend data reviewed for this assessment indicate that concentrations of contaminants are not increasing in sediments, mussels, or fish in the open coastal zone of the bight. Taken in total, data reviewed for this report suggest that where they were once high, concentrations of most contaminants in the open coastal zone have been decreasing in sediments and mussels (Table 1). However, we are less certain about trends in bays and harbors.

Sediments. Where adequate time series or dated sediment core data exists, the data show that where concentrations of metals, DDT, PCBs and PAHs were once elevated, they have been declining (Table 1). This includes sediments in offshore basins (Santa Barbara, Santa Monica, San Pedro) and at some sites along the coastal shelf, including Santa Monica Bay, Palos Verdes, and at the Orange County outfall. Declines of levels of many contaminants (arsenic, cadmium, chromium, copper, mercury, zinc, PCBs, and DDT) in sediments from Palos Verdes have followed declines in emissions of these contaminants from the nearby sewage outfall [9]. Decreases in lead and PAHs have also been noted in sediments from some areas. Unfortunately, there is inadequate information on temporal trends of selenium, silver, tin, chlordane, and dieldrin in sediment of the Southern California Bight.

Mussels. Again, where adequate time series exist, the data show that where concentrations of metals, DDT, and PCBs were once elevated, they have decreased since the 1970s. These trends are inferred largely from reconstructions at two coastal sites—Royal Palms (White's Point) on the Palos Verdes Peninsula and at Oceanside (Table 1). During the period 1977 to 1986, concentrations of DDT, PCBs, chromium, and lead clearly declined by factors of 2 to 10 at Royal Palms but were variable and at lower concentrations at Oceanside. The National Pesticide Monitoring Program fortunately captured peak levels of DDT between 1967 and 1969 in mussels from three widely scattered locations (Point Mugu, Anaheim Bay and Hedionda Lagoon [15]. Most of the decrease of DDT and PCBs in mussels had already occurred between 1971 and 1974 commensurate with effective source control in municipal wastewaters. However, major decreases in emission continued through the 1980's (to 1 per cent of 1974 inputs) but this was not reflected in further dramatic declines in coastal mussels. Indeed, mussels at Palos Verdes, Corona del Mar and Oceanside actually experienced increased levels of DDT and PCB's between 1979 and 1983-84 suggesting there were active sources other than the municipal discharges at that time [1]. Further, although lead also decreased with decreased emissions from wastewater outfalls, strong gradients radiating out of the Los Angeles area remain, suggesting that non-point

Table 1

Changes in concentrations of contaminants in sediments and mussels, and in mass emmission from six major wastewater discharges, at selected sites since early 1970's or as otherwise noted. Values are fraction of initial-year concentrations computed for significant trends. Sediment core data not yet subjected to significance tests. Data from several sources described in Mearns et al [1].

	Trend from Offshore Sediment Cores			Mussels 77-86 Fraction of 77		Total Wastewater Emissions, 1974-88 Fraction of 1974
Contaminant	St. Barbara Basin	Santa Monica Basin	Palos Verdes	PV	Oceanside	
Arsenic	No data	No data	Decrease	No data	No data	0.43
Cadmium	Decrease	Decrease	Decrease	No change	No change	0.07
Chromium	Decrease	Decrease	Decrease	0.24	No change	0.04
Copper	No change	Decrease	Decrease	No change	No change	0.13
Mercury	No data	No data	No data	No Change	No change	0.17
Lead	Decrease	Decrease	Decrease	0.04	0.33	0.24
Selenium	No data	No data	Decrease	No data	No data	0.39
Silver	No data since '71	No data	No data	No change	No change	0.50
Tin	No data	No data	No data	Increase (?)	No data	No data
Zinc	Decrease	Decrease	Decrease	No change	1.8(not tested)	0.12
PAH's	No data	Decrease ('60's , '70's)	No data	No data	No data	No data
DDT	No data since 71	No data since 71	Decrease	0.40	No change	0.01
PCB	No data since 71	No data since 71	Decrease	0.33	No change	0.01
Chlordane	No data	No data	No Data	0.25 (Long Beach)	0.43	No data
Dieldrin	No data	No data	No Data	Peak '83	Peak '83	No data

sources dominated by the mid-1980's. Chromium was the only metal that clearly and unequivocally decreased in response to decreased wastewater emissions at Palos Verdes (but on a very local scale).

In contrast, concentrations of silver, copper, mercury, and zinc were variable in Royal Palms and Oceanside mussels with no obvious long-term upward or downward trend. This lack of a trend does not follow the decreases in many of these metals seen in sewage effluent emissions and in sediment (last column, Table 1). Concentrations of these four trace elements were, and continue to be, higher in mussels from Royal Palms than from those at Oceanside.

A rather startling additional observation, evident especially in the NS&T data, is for cadmium, which has been increasing at Royal Palms but not at other sites, since 1986. Since 1986 there has been increasing as inputs from sewage have decreased. These observations suggest that, unlike chromium, lead, DDT, and PCBs, actual concentrations and inter-annual variations of cadmium, copper, mercury, and zinc in mussels at this site are independent of sewage inputs.

Another interesting pattern occurred for mercury: Concentrations in mussels at three long-term monitoring stations (Oceanside, Palos Verdes and Catalina Island) increased dramatically in the early 1980's then decreased just as dramatically [1]). The net effect of this peak was to squelch the possibility of discovering any long-term (decadal) trend. This is unfortunate since no other substrate (sediments, macroinvertebrates, fish) have been consistently monitored since the mid-1970's. In this connection, it is important to note that the highest concentrations of mercury in mussels has occurred at San Miguel Island where Flegal et al. [21] postulated that the major source was from pinniped excrement. Inadequate information exists to evaluate temporal trends in mussels for tin and selenium.

Fish and Shellfish. A result of work by MacGregor [22] and Stout and Beezhold [23] there is strong evidence for a chronic rise of DDT contamination in both inshore and offshore pelagic fish throughout the 1960's. There is equally strong evidence that concentrations of DDT and PCBs have declined at least ten-fold since the early 1970's in fish and shellfish of the coastal shelf. However, some species, such as the white croaker (*Genyonemus lineatus*) continued to maintain unacceptably high concentrations during the 1980's at various sites between Long Beach and Santa Monica.

There is inadequate information to determine trends in concentrations of metals, PAHs, chlordane, and dieldrin in fish. However, since there is evidence that metals have not accumulated in fish and, in some cases, they have been depressed, there is little reason to expect concentrations to decline with time. Indeed, for some metals (cadmium and possibly arsenic) there may be reason to expect concentrations to increase in fish from areas such as Palos Verdes and San Diego Harbor. Many species of fish and other seafood invertebrates have been measured only once or twice during the past two decades. Without repeated surveys, it is impossible to judge the extent to which trends in inputs, sediments, and mussels are also reflected in the majority of important resource species.

Another major problem is that most of the long-term trend data are from the open coast. There are major gaps in the trend monitoring data for *bays, harbors, and lagoons* where long-term monitoring has been virtually nonexistent. An exception is Marina del Rey where PCB concentrations in sediments may be increasing [24]. This is consistent with an analysis of NOAA NS&T mussel data that suggests that while there have been recent (since 1986) major decreases of PCB contamination at offshore and island sites, concentrations in mussels collected at harbor entrances ave remained unchanged or are increasing [1]. In contrast, sediment metal concentrations in the Rhine Channel of Newport Bay decreased dramatically between 1981 and 1988 following controls on vessel cleaning operations [25]. However, as of 1986, this remained the most contaminated site on the coast with respect to total metals (see below).

STATUS: WHERE ARE THE "HOT SPOTS"AND WHERE ARE THE CLEANEST AREAS?

A major problem is defining the term "status." Even though most of the areas reviewed in this report were surveyed more than once, most were not surveyed at the same time (during the same years). There is no "base" year on which to compare most of the 73 sites of interest. However, most were surveyed at least once between 1978 and 1986 (data collected after 1987 were not reviewed in depth for this report). Thus, it must be borne in mind that the conclusions about "status" in this report are focused primarily on the period 1978-86. They could easily be outdated by a new assessments.

Sediments. It is instructive to bring "recent" contaminant data together in one, albeit incomplete, framework. For 14 sites with fairly complete and comparable data, I computed the extent to which median concentrations of five trace metals exceeded the best estimates of natural or background concentrations (as described in [1]) then group together the ranges of excess over background concentrations and ranked them in descending order by their grand medians. As shown in Table 2, the Newport Bay shipyard area (Rhine Channel) was the top metal "hot spot" of the mid-1980's, on average 10.4 times more contaminated than background (range 5.2 times for chromium to 55 times for copper). The five most contaminated areas, where on average metals exceeded background concentrations by a factor of 4 or more, included all four harbor areas (Newport shipyards, San Diego, Marina del Rey and Los Angeles/Long Beach Harbors) as well as the Palos Verdes shelf. By contrast, two undeveloped lagoons, two open coastal outfall sites and one open coastal area had small increases in metal contamination (grand medians 1.6 to 2.6 times background). Finally, the least contaminated regions included most of the coastal shelf between Point Conception and the United States-Mexico international border, including sewage outfall sites at Oxnard (near Port Hueneme) and Orange County. A more detailed assessment of increase ratios indicated that Point Loma and Orange County 60m deep sites exceeded the median of background on the order of a factor of two, and these excesses

Table 2

Site ranking of median increase ratios of concentrations of five trace metals at 14 southern California sites in recent years. Metals and background concentrations (ppm dw) are: cadmium (Cd), 0.4 ; chromium (Cr), 29 for coast and 6 for bays; copper (Cu), 10; lead (Pb), 10; and, zinc (Zn), 44. Based on data from various sources compiled in Mearns et al [1].

Site	Year	Grand Median	Range of Medians (metal) Min	Max	Environment
Newport Bay Shipyards	1986	10.4	5.2 (Cr)	55 (Cu)	Harbor
Palos Verdes Shelf (60m)	1985	9.4	6.9 (Pb)	27.5 (Cd)	Open coast, LA County outfall
San Diego Harbor	1983	8.4	4.5 (Cd)	18.8 (Cr)	Harbor
Marina del Rey	1985	6.8	1.2 (cd)	13.3 (Cu)	Harbor .
LA-LB Harbors	1978	4.2	1.5 (Cd)	14.5 (Cr)	Harbor
Santa Monica Bay 60m	1985	2.6	0.8 (Zn)	8.8 (Pb)	Open coast, LA City outfall
Upper Newport Bay	1980	2.6	2.1 (Cd)	4.8 (Cr)	Lagoon
Tijuana Slough, south arm	1988	1.8	1.3 (Cu)	2.0 (Cd, Pb)	Lagoon
Point Loma shelf 60m	1985	1.7	0.7 (Cr)	2.7 (Cd)	Open coast, Point Loma outfall
Newport to Dana Point 60m	1985	1.6	0.3 (Cd)	1.7 (Cu)	Open coast
Orange County Shelf 60m	1985	1.2	0.8 (Cr)	1.8 (Cu)	Open coast, Orange County outfall
Pt Hueneme to Pt Dume 60m	1985	0.9	0.5 (Cd)	1.1 (Cr, Cu)	Open coast
Santa Barbara Shelf 60m	1985	0.7	0.4 (Pb)	1.1 (Cr)	Open coast
Tijuana Slough, north arm	1988	0.5	0.23 (Cu)	1.1 (Cr)	Lagoon

were dominated by silver [1]. The remainder of the surveyed coastal shelf (Newport to Dana Point and Point Dume and across the Santa Barbara shelf) had no excesses of contamination greater than 1.7 and some were below average background values. The least contaminated coastal regions have been the Santa Barbara shelf and the shelf between southern Orange County and Point Loma.

Intermediate in rank between these extremes is Santa Monica Bay. Excluding a now-abandoned sludge discharge zone (100m deep), the shelf of central Santa Monica Bay has experienced metal contamination at levels 2 to 3 times background; it also appears to be intermediate on a gradient of DDT contamination originating at Palos Verdes.

An unanswered question is the extent to which the more contaminated sites were contaminated at potentially toxic levels. I was unable to devote the resources for a detailed review of this question. However, I can point out to the reader the range of possible answers. For example, in all areas DDT concentrations, even where they were lowest such as in San Diego Bay, exceeded the low effects range (ER-L) of 0.005 ppm computed by Long and Morgan [26]: however, only two areas (Palos Verdes and Bolsa Bay) had sediments exceeding the median effects range (ER-M). At the other extreme, DDT concentrations did not exceed Long and Morgan's ER-M for arsenic or PAHs, and sediments at only one region, Palos Verdes (1985) exceeded the ER-M for chromium.

Considerable caution is in order here. First, this is a far from complete list of sites and areas: it is especially missing bays and lagoons such as Mission Bay, Batequitos Lagoon, and marinas of Ventura County. Data from these areas has either been overlooked in this review, or it does not exist. Second, there has been no synoptic survey of Los Angeles Harbor since 1978, nor of Upper Newport Bay since 1980. Third, as of the mid-1980's, there are important gaps in the data. For example, there is no data for silver (Ag) for bays and harbors, nor recent data for mercury (Hg) in less-urbanized coastal areas such as the Santa Barbara or Hueneme shelf, or even at the Point Loma outfall site. Lack of data from these possible reference areas puts a serious constraint on interpreting data from the urban and outfall areas.

Mussels. As noted previously in individual chapters of [1], and in the discussion below on trends, only some contaminants in mussels (such as DDT, PCBs, and chromium) appear to reflect the same contaminant patterns as revealed by sediments. However, mussel surveys have identified additional areas of exceptional contamination by one or more materials; these include, for example, Oceanside Harbor (copper), Bolsa Bay (lead, chromium, and, possibly DDT), and, in the past, Upper Newport Bay (DDT). Detailed review of other hot spots revealed by mussel surveys are given in the full report [1].

Fish and Other Shellfish. With the possible exception of tin (Sn), there is no indication of excess trace metal contamination in tissues of fish from any adequately-sampled site. However, there are clear indications of metal depletions or depressions in fish from several areas. On this point, the geographical patterns of contamination in fish livers from the 1984 NS&T Benthic Surveillance Program are in good agreement with historical patterns. That is, most of the metals reviewed here are elevated in livers of fish from sites presumably remote from major metal inputs (such as Dana Point). They are low or even possibly depressed in livers of fish from sites, known from other measurements, to be contaminated with metals, PAHs, PCBs, and DDT (sites in LA/LB and San Diego harbors). Although not tested statistically here, the agreement is surprising and suggests that a phenomenon of regional scale, leading to metal depletion, has been in effect for nearly two decades. As suggested in [1], there is some indication that high levels of organic contaminants may be causing depressed levels of some metals in fish. A consequence of these patterns is the possibility that continued waste management activities in the Los Angeles area and, to some extent, San Diego Harbor, may lead to decreasing levels of PCBs, DDT, and tin but increasing levels of other trace elements in livers of nearshore bottom-fish with concentrations varying by factors of about 2 to 4. Exceptions to this pattern include DDT, PCBs, chlordane, and total tin. That is, in NS&T and/or other programs, it is clear that these contaminants are elevated in fish livers in approximate proportion to levels in nearby sediments, sources, or in mussels. Again, statistical tests would resolve exception, but this appears to be a major pattern common to both NS&T and regional data sets.

CONTAMINANTS OF CONCERN?

Which contaminants are of concern, which are not, and which have too many uncertainties for resolution? Table 3 is a short-hand version of what I have learned about a number of contaminants in the Southern California Bight. Various contaminants are listed down the left hand side. Across the table, from left to right, various kinds of information about each contaminant are checked. The first column indicates if the contaminant occurs somewhere in excess concentrations in sediments. To the right I indicate the extent to which each contaminant is also found in excess in various well-monitored marine organisms, and whether that contaminant has occurred in sediments in toxic concentrations. Finally, I note, from all available information, the extent to which inputs and concentrations are getting worse (increasing). This kind of summary helps determine which contaminants are of most and least concern now, and, perhaps, which need more information.

As seen in Table 3, all contaminants or contaminant groups have accumulated in sediments in at least one locality in the bight. However, not all contaminants have accumulated in mussels, macroinvertebrates, and fish. Only three contaminants (mercury, DDT, and PCB) showed clear evidence of biomagnification. Finally, existing evidence, where available, indicates that concentrations have been decreasing for 14 contaminants while the direction of trends was increasing for one (cadmium at Palos Verdes) and remains uncertain for another (silver). Below, I review attributes of these in order of their final ranking.

Anthropogenic Contaminants of Continuing Concern (DDT, PCB, PAH, Chlordane). Four organic chemical classes are of continuing concern and should be subject to

Table 3

Summary of patterns of contamination in the Southern California Bight. "No" responses are intentionally left blank.

Contaminant		Excess Accumulation in:						
	Sediment?	Mussels?	Invertebrates?	Fish?	Biomagni-fication?	Public Health Concern?	Increasing in Sediment or Biota?	Level of Concern
DDT	Yes	Yes	Yes	Yes	Yes	Yes		1
PCBs	Yes	Yes	Yes	Yes	Yes	Yes		1
Chlordane	Yes	Yes	no data	Yes	?	?		1
PAHs	Yes	Yes	Yes			?		1
Mercury	Yes	Yes			Yes	Yes		2
Arsenic	Yes					?		2
Tin (Organo)	Yes	Yes	?	Yes	?	?		3
Dieldrin	Yes	Yes	?	Yes	?			4
Cadmium	Yes	Yes	Yes				Yes	3
Silver	Yes	Yes	Yes				?	3
Lead	Yes	Yes	?					3
Chromium	Yes	Yes	Yes					4
Copper	Yes	?	Yes					4
Selenium	Yes	?	?					4
Zinc	Yes							4
Volatile organics	Several							4

Levels of Concern: 1 - Anthropogenic contaminants of continuing concern; 2 - Natural chemicals in seafood but not elevated by anthropogenic sources; 3 - Contaminants of uncertain concern; 4 - Contaminants apparently not of concern. Concern based on bioaccumulation, not concentrations in water. Modified from a similar analysis in [1].

continued or increased surveillance. These include: PAHs, PCBs, DDTs, and chlordane. The organic chemicals have occurred in the bight in marine organisms at concentrations of concern in terms of seafood quality or possible toxicity to marine organisms.

The extent of past DDT contamination of sediment and fish was incredible. In no other marine or coastal area of the United States have DDT concentrations in fish reached levels that have occurred historically in the Southern California Bight. DDT was an important region-wide contaminant of fish for at least a decade (1970-80). For example, concentrations of DDT in white croaker from Palos Verdes, which resulted in fishery advisories in 1985, once occurred 100 miles to the south near San Diego [23]. These high levels were responsible for the near extermination of brown pelicans [27] and for the death of zoo seabirds fed 73 locally caught fish [28]. Had the 1985 level of concern been applied in 1970 and 1971, it is likely that the entire coast of southern California and beyond would have been posted with seafood consumption advisories. However, the most recent data suggest that the widespread occurrence of high levels in fish flesh is now restricted to the immediate coastline of the Palos Verdes Peninsula in large or older fish and in bottom fish in immediate contact with contaminated sediments.

DDT concentration apparently increases with trophic level. Highest levels have occurred in sharks and bottom fish. Concentrations in various fish appear to have decreased dramatically since the early 1970s. The principal reason for the sharp decrease is apparently control of emissions from the JWPCP outfalls at White's Point (Royal Palms) off Palos Verdes. Although known inputs are now negligible, DDT remains in some species of fish from the Palos Verdes area, San Pedro Bay, and Santa Monica Bay. Concentrations may not appear to decrease as rapidly in large, long-lived fish, such as large kelp bass, as in younger kelp bass or in short-lived small species, such as perch. Therefore, it is not only important to continue monitoring DDT in sportfish from the Palos Verdes and appropriate reference areas, but to develop monitoring protocols that allow for prediction by age class or age group.

PCBs have accumulated to potentially toxic levels in sediments from outfall areas such as Palos Verdes and bays and harbors such as Santa Monica Bay, San Pedro Bay, and San Diego Harbor. They are widespread contaminants of mussels and fish also, and recent assessments attribute most risk of cancer from consumption of contaminated seafood to PCBs. However, most evidence suggests that high levels of PCB contamination of fish have been more localized than DDT contamination in the Southern California Bight. Fish from Santa Monica Bay, Palos Verdes, San Pedro Bay, and San Diego Harbor have clearly been highly contaminated. In addition, PCB contamination increases with trophic level and highest levels of PCBs have been seen in sharks and bottom fish. Concentrations are lower now in fish from the Los Angeles area than in past years, but evidence is lacking to determine if concentrations have been decreasing or increasing in San Diego Harbor or have changed throughout the bight as a whole. Also lacking is adequate data from other bay, harbor, and inshore areas (especially

Port Hueneme, Marina del Rey, inner San Pedro Bay, Huntington Harbor, Newport Bay, and the lagoons of San Diego County).

A possible impediment to tracking declining PCB levels in coastal fish and macroinvertebrates is the high detection limit (0.2 ppm ww) for PCBs imposed in monitoring programs of some dischargers. For example, muscle of most fish sampled in the Orange County monitoring program contain PCBs at concentrations below this detection limit. In the past, detection limits were lower (0.001 to 0.01 ppm ww). If detection limits are not soon reduced to their earlier values, there will be no useful record of reduction due to continuing source control.

Since they are potent carcinogens, PAHs are of concern wherever they are found. In the data reviewed here for the Southern California Bight, one or more higher molecular weight PAHs (such as benzo(a)pyrene) have been measured in sediments, mussels, and/or fish from several areas: Santa Monica Bay near the now abandoned sludge outfall; San Pedro Bay and the adjacent Palos Verdes shelf and San Pedro Basin; in San Diego Harbor; and in several smaller harbors. Beyond these sites, along the open coast and in the few island sites that have been surveyed, PAHs do not accumulate in sediments or mussels. Although PAHs do not accumulate in fish, measures of PAH metabolite compounds in bile of benthic fish from several harbor areas of the Southern California Bight have shown that fish are exposed to PAHs. Temporal trends in PAH levels remain uncertain, but may have declined.

Appropriate measures of PAHs or their metabolites should be continued to determine if concentrations are increasing or decreasing. Additional harbors and urbanized embayments should be sampled, especially Newport Bay. It is critical to continue to monitor the same recently sampled sites, species, and substrates.

Concentrations of chlordane in sediment from Marina del Rey, Palos Verdes, and San Diego Harbor have exceeded those that are potentially toxic to sensitive marine species. Clearly chlordane compounds have entered marine ecosystems of the bight. Chlordane may biomagnify. Although concentrations in edible tissues have not exceeded FDA guidelines, some were close. Excesses may have occurred in previous years (early 1970's) in fish near the Los Angeles area. Many areas have not been surveyed for chlordane in fish or other seafood organisms, but data from mussels in Marina del Rey suggest that fish from all urban harbors should be measured. Long-term trends in chlordane concentrations remain unknown. Since chlordane is apparently toxic to sensitive species at very low concentrations, detection limits for sediments should be correspondingly low.

Natural Chemicals of Concern in Seafood but not Elevated in Seafood by Anthropogenic Sources (Mercury and Arsenic). Two materials, arsenic and mercury, have also occurred at concentrations of concern in terms of seafood quality, but high concentrations are apparently not the result of known and/or controllable anthropogenic sources. Here the only realistic management action is to make sure we understand the public health risks, or non risks, and advise accordingly.

Mercury has accumulated in sediment from several areas (especially at Palos Verdes, Newport Bay, and Marina del Rey) at concentrations that may have been toxic to some benthic invertebrate species. Mussels from Marina del Rey have contained elevated levels of mercury. Mercury occurs naturally in high concentrations in fish from many areas and there is strong evidence that some species contain mercury much higher than the FDA action limit of 0.5 ppm ww. Mercury increases with trophic level. For example, all sharks and some fish yielded one or more samples of flesh at or far above (up to 20 times) the FDA limit. Although mercury in sediment has declined in response to source controls, temporal trends in mussels and fish remain uncertain. Since the highest levels of mercury were seen in fish from areas located far from known sources, it does not appear that mercury from coastal waste discharges is responsible for concentrations observed in fish. Thus, the only management action possible is to conduct adequate surveys of large fish and sharks and warn consumers as appropriate. It would be useful to conduct mercury surveys to identify species and sizes of fish, sharks, and rays that meet federal or local guidelines.

Although accumulations of arsenic in sediment have not reached potentially toxic levels, arsenic occurs naturally in high concentrations in selected species of mollusks, fish, and crustaceans. Arsenic concentrations in certain fish are sufficiently high to cause concern from a consumer standpoint. Arsenic concentrations in flatfish, particularly Pacific sanddab, are high, especially in areas remote from urban pollution sources. Concentrations are depressed in some species from polluted areas. Arsenic concentrations in sediment have declined, but no change has been noted in mussels or fish. The source of high levels of arsenic in certain species in unknown, but it does not appear to be from anthropogenic sources and may, like mercury, be a natural phenomenon. It is thought that arsenic in fish and shellfish exists in the less-toxic organic form, but the risk to human health from consuming these concentrations of arsenic is unknown. Accordingly, the only effective management action is to survey arsenic adequately in fish from the bight and post advisories as appropriate.

Contaminants of Uncertain Concern (Organotin, Tin, Cadmium, Silver and Lead). There are five contaminants of uncertain concern, each for a different reason.

Organotin compounds are clearly contaminants in sediments of San Diego Harbor, San Pedro Bay, Marina del Rey, Oxnard Harbor, and Palos Verdes. Levels of tin in mussels have been highest in Newport Bay and San Diego Harbor. Highest levels of tin have been seen in fish from San Diego Harbor although trends with time have not been well investigated. Surveys should expand into other marinas and a concerted effort should be made to measure organotin concentrations in livers of fish from these areas. At the same time, data should be obtained on total or inorganic tin concentrations to quantify any consistent relationship to organotins.

Cadmium has accumulated in sediments and may have occurred at toxic levels at Palos Verdes. Elevated concentrations of cadmium have been measured in San Diego Harbor. Cadmium does not appear to accumulate in fish or undergo biomagnification. However, 15 years ago several macroinvertebrates from Palos Verdes have contained higher levels (factor of two above reference sites) of cadmium than those from more remote areas.

There are two enigmas with cadmium. First, cadmium is recently increasing in mussels at Palos Verdes despite the fact that inputs have been decreasing. Second, mussels and fish from islands and remote sites in the Santa Barbara area and from remote areas of Baja California contained concentrations two to ten times higher than those from the urban areas. Arsenic also shows this behavior, but not as dramatically. Third, there is an inverse relationship between concentrations of cadmium and DDT in mussels. These observations suggest cadmium from anthropogenic sources has not been available to the biota on a regional basis and that natural sources, coupled with some kind of inhibition of uptake caused by DDT, are two factors causing *depressed* levels of cadmium in urban marine biota.

Silver has accumulated in sediment from many areas and may have existed at toxic levels at Palos Verdes and Santa Monica Bay. Results of silver analyses show a meso-scale gradient in mussels occurred along the coast from central California to northern Baja California. The presumption is that silver inputs from sewage have, like DDT, spread over large areas of the coast. The dilemma is that unlike DDT there is evidence that silver is not accumulating in edible tissues of most seafood organisms. However, fish from truly remote areas of Baja California or central California have not been measured so it may be premature to conclude a gradient similar to that in mussels does not also exist in fish. Levels of silver in sediment and mussels have apparently not changed with time. Trends in fish are unknown.

Lead has clearly contaminated sediments and mussels of the Los Angeles, Orange County, and San Diego County coastal areas for many years. Concentrations of lead in sediments of Palos Verdes, Newport Bay, San Diego Harbor, and Marina del Rey may have been high enough to cause toxicity in sensitive species. In the 1970s, concentration gradients in mussels reflected the importance of aerial fallout from auto emissions as a major source. Concentrations in mussels declined during the late 1970s but have leveled off and remained unchanged during the 1980s despite continued reductions in sewage effluents and resultant decreases in sediment concentrations. Concentrations are still as much as 5 times higher in mussels from Palos Verdes than in those from Oceanside, suggesting continued inputs from the Los Angeles area, presumably from aerial fallout. Urban runoff and deposits in flood control channels may be major potential sources [29]. New measurements of lead (using ultraclean facilities) in tissues of fish and macroinvertebrates are needed to determine if concentrations have changed since the late 1970s. Most existing recent tissue data (except from mussels) is invalid for this comparison unless it can be demonstrated that samples were prepared and analyzed in a lead-free environment. Public health implications of elevated lead in mussels from urban coastal areas should be addressed.

Contaminants Apparently of Minor Concern (Chromium, Copper, Selenium, Zinc, Dieldrin, Volatile and Extractable Compounds). Existing evidence indicates that four metals and large numbers of organic chemicals undergo neither bioaccumulation nor biomagnification in fish of the bight, or that elevations in macroinvertebrates are, or have been, extremely localized and not of regional significance. Chromium, copper, selenium and zinc do not accumulate in fish or most macroinvertebrates in the bight, and none undergo bioaccumulation. None have occurred at levels of public health concern in sea food. Tissues of several species of shellfish at Palos Verdes were once slightly (2-fold) contaminated with these metals. It might be worthwhile to resample them once at Palos Verdes to confirm that concentrations have decreased commensurate with known reductions in emissions. This was a local, not a regional phenomenon. On the other hand, all have been subject to intensive industrial source control efforts which have resulted in decreased levels in sediments but no change in biota (where they have been monitored). One, selenium, was actually depressed in several species from otherwise contaminated areas (similar to cadmium, as discussed above). Levels in bays originating from urban runoff may be of local concern (e.g. zinc from tires [30] and lead in flood control channels [29]).

The organochlorine pesticide dieldrin may have once been a contaminant of concern in the bight, but existing data, although sparse, does not indicate important concentrations of dieldrin in any substrate.

Data on several dozen volatile organic compounds measured during the past 5 years as part of the Orange County Sanitation Districts monitoring program, indicate that none of these compounds have accumulated in fish or shellfish. In the past, several chlorinated benzenes were elevated in macroinvertebrates near the deep-water outfalls in Santa Monica Bay. Partition coefficient studies have demonstrated low potential for bioaccumulation. The lack of bioaccumulation of most volatile and extractable organic chemicals in fish and macroinvertebrates of the bight is not surprising due to their low partitioning coefficients and volatility. However, many may still be important as toxicants in water and at the sea surface in local areas of heavy inputs. Phenol was not a target chemical for this review, but may be important since quantifiable concentrations appeared in fish from the Los Angeles and Orange County areas during surveys conducted in the early 1980's. A review of phenol inputs and distribution may reveal a need for additional measurements.

CONCLUSION

More than 18 million dollars have already been spent annually on ocean monitoring in the Southern California Bight during the past two decades [2]. While there are many incompatibilities among the contaminant surveys, they nonetheless, represent a remarkable monitoring achievement and a record of contamination and recovery that is continuing today.

For all of the chemicals reviewed here, concentrations in sediments of the open coast have either not changed (i.e., were never elevated) or have declined dramatically since the early 1970's. Where they were once elevated on the open coast, concentrations of many contaminants in mussels and fish have also declined (exception is cadmium in mussels at one site). Our major uncertainty is what is happening in bays and harbors which have been, or contained, the most contaminated sites in the bight. This is not surprising considering their proximity to urban, industrial, and residential areas and their restricted circulation. Biological effects surveys can most fruitfully be focused on these areas.

The least contaminated areas for sediments are the coastal shelf areas near Santa Barbara and between southern Orange County and Point Loma. The now relatively low concentrations of contaminants in sediments and mussels near open coastal discharge sites such as at Point Loma and Orange County suggest that existing source control wastewater treatment are effective.

Many specific recommendations to improve regional monitoring and assessments are given in the full report of this project [1]. These include improvements in species and site selection, in chemical assessment methods (such as appropriate detection limits) and increased emphasis on field toxicology assessments. Monitoring should be continued and coordinated [2] and much more emphasis put on bays and harbors. Existing data should be made available to all agencies through some type of regional archive and should also be used to determine how best to continue or build upon existing and historical time series so they will provide points of reference for marking the further progress of reductions in contaminant loading. There is urgency in collecting existing data since many of the technicians and scientists who have provided much of these data will soon retire and may not be accessible for evaluating data quality (some have already passed away).

ACKNOWLEDGEMENTS

I thank the seven co-authors of the full report [1] who conducted most of the work of this assessment under my guidance: Michael Buchman, Joesph Golas, Howard Harris, Gunnar Lauenstein, Donald MacDonald, Mary Matta and Gary Shigenaka. But I also would like to thank perhaps nearly 100 California and national scientists and technicians who provided data, answered numerous questions and provided critical review comments during the course of this project. The opinions expressed here are those of the author and do not necessarily represent the official position of NOAA.

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Marina Del Rey As A Fish Habitat: Studies Of The Fish Fauna Since 1977

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Abstract. Marina del Rey is an enclosed small boat harbor connected to the ocean by a one mile entrance channel. It differs from adjacent semi-enclosed harbors such as King Harbor, Santa Monica and Los Angeles-Long Beach Harbors by access to the longshore current and patterns of circulation. The entrance breakwater, though small and shallow, compares favorably to other rocky breakwaters in Santa Monica Bay. The remainder of the marina is 12 feet deep at mean tide with a silt bottom and little hard substrate other than vertical retaining walls, piling and floats. There is one exception; Basin D terminates in a sandy, swimming beach with some "sea grass" beds. We have surveyed the fish assemblage within Marina del Rey, at least semiannually in 1977-79 and since 1984. More than 90 species or higher taxa have been identified. A mean of 40 species (SD = 45) were observed in the surveys. Our limited sampling indicates considerable annual fluctuation in both adult immigration and juvenile recruitment to the marina. The shallow swimming beach in the marina appears to serve as a refuge for Ballona wetland species. The absence of shoaling (sloping) shallows in most of the marina may inhibit this function. Even though Marina del Rey has poor circulation, is subject to pulses of non-point source pollution, and has high summer temperatures, it serves as a regular habitat for a diverse ichthyofauna.

MARINA DEL REY

Marina del Rey, the largest manmade marina in the world with more than 6000 berths, is located in central Santa Monica Bay (Fig 1). The marina was constructed in the early 1960's on a former wetland that had been degraded by dumping, filling, oil extraction, agriculture, and urbanization of the surrounding area of greater Los Angeles. The marina connects to Santa Monica Bay by a mile long Entrance Channel which includes a terminal breakwater parallel to the shore and two lateral breakwaters bordering the channel which extended seaward about 1000 feet. The marina includes the Main Channel and eight boat basins. At the terminus of Basin D there is a sandy swimming beach, while at Basin E the farthest from the entrance, there is a tidal gate connected to the Oxford Flood Control Basin. A tide gate in the Entrance Channel serves Ballona Lagoon and the Venice canal system. Large storm drains also enter Basins G and H. The Ballona Creek Flood Control Channel parallels the Entrance Channel to the south and may affect the quality of the water in the entrance channel, or even the entire marina if significant quantities of storm waters flow or inland spills occur.

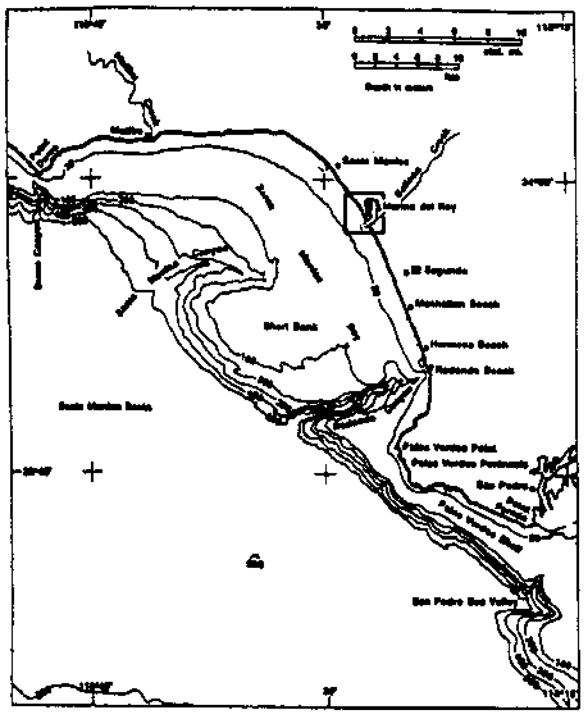

Figure 1. Location of Marina del Rey within Santa Monica Bay.

Water entering the marina may derive either from the California Current from the northwest or from the Southern California Eddy flowing from the southwest. This latter is an offshoot of the California Current, the eastern boundary current of the North Pacific Gyre. The eddy is the major influence on the Southern California Bight, whose waters are considerably warmer than those north of Pt. Conception, where waters are under more direct influence of the California Current. Southern California is also a site of seasonal upwelling which brings higher nutrient water from the undercurrent and benthos to the surface. In winter, the Davidson Countercurrent may bring tropical waters northward, sometimes surfacing in Santa Monica Bay as part of the upwelling in the gyre.

In some years, the tropical flow may continue northward for longer periods, producing the local El Niño-Southern Oscillation (ENSO) phenomenon. Sewage discharges and power plant cooling discharges also affect the Southern California Eddy as it circulates to the north in Santa Monica Bay.

STUDIES AT MARINA DEL REY

Harbors Environmental Projects at the University of Southern California performed the first baseline environmental studies of the marina under joint USC Sea Grant and County of Los Angeles funding in 1977-1979, during which a variety of fish sampling techniques were tested, along with sampling for water quality, nutrient chemistry, sediment chemistry, phytoplankton productivity, zooplankton and benthic organisms [1,2]. Studies were resumed in 1984 under sponsorship of the county, with the *Vantuna* Research Group performing the ichthyological surveys (Figure 2).

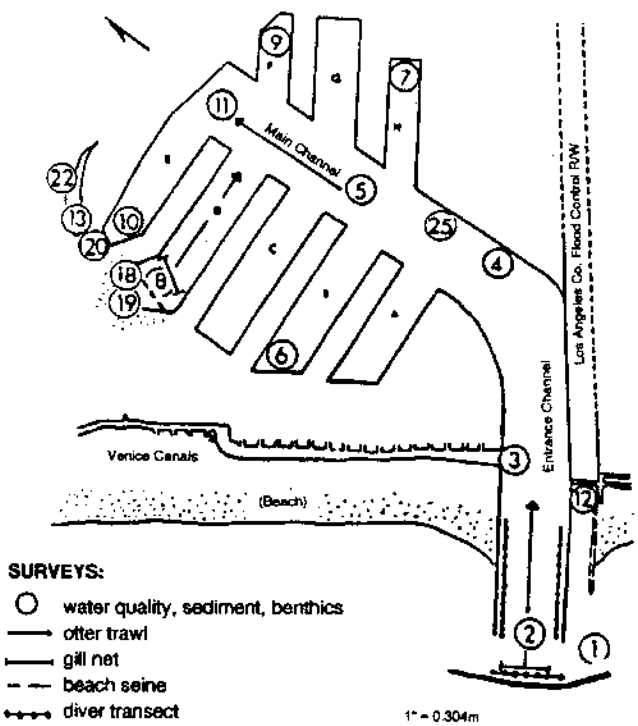

Figure 2. Survey stations in Marina del Rey.

Ichthyological samples have been taken twice a year, in October and May since 1984 [3-8]. The methods during this period have been standardized and are comparable: diver surveys at the breakwater (three 10-minute swims); 15 ft otter trawls towed for 10 minutes at three stations; three minute ichthyoplankton sample tows (meter nets) at surface and bottom; three one-hour sets with a 100 ft x 6 ft variable mesh gill net; and a 100 ft x 6 ft 1/4in mesh beach seine sample. All work was done during the day. No poison or quinaldine stations have been carried out during this period and, therefore, probably some four to six cryptic rock reef species will have escaped our analysis.

FISH FAUNAL COMPONENTS

The fish fauna of Santa Monica Bay is typical of the Southern California Bight and is made up of inshore species belonging to the San Diegan Warm Temperate fauna. Occasionally elements from the Cold Temperate Oregonian (Californian) and the tropical Panamic faunas may be present. During ENSO events, more tropical elements appear, while during cold anomalies temperate elements can be abundant.

An additional factor of importance when discussing fishes of marinas is the loss of lagoonal and esuarian habitats (wetlands) along the Southern California coast. Most natural wetlands have been long since developed as harbors and marinas, or filled for terrestrial development. Elements of their endemic faunas have refuged in marinas where they may have survived.

SPECIES COMPOSITION AND NUMBERS

Since 1977, a total of 81 species have been identified from a total of 90 taxa (some not idendifiable to species level) in our studies of Marina del Rey. This compares to 153 species identified at King Harbor in more extensive studies conducted since 1974 [9]. King Harbor, located at the southern end of Santa Monica Bay, between Hermosa Beach and Redondo Beach, is entirely constructed from breakwaters and its fish fauna is typical of marine rocky shore and sand beach. Marina del Rey, by contrast, as an inland facility, is dominated by estuarine or wetland species except at the Entrance Channel breakwaters. Even with the much larger species list at King Harbor, 12 species recorded at Marina del Rey are not present in King Harbor as adults (four have been taken as larvae). Marina del Rey thus serves as a refuge for wetland species as well as a resident site for typical bay forms. Thirty-eight of the 81 species in Marina del Rey fit this category with the remainder species commonly taken at the entrance breakwater.

Of the 81 species observed by these standard methods since 1984, 40 (51%) are generally fishes requiring or commonly occurring on soft substrate, the remainder are rock reef species or epipelagic. This contrasts with King Harbor where 50 (42%) of the 120 species observed by divers are soft substrate forms.

We considered that species we observed in at least four of the last five years of this standardized study are residents of Marina del Rey, while those that are present two to three of the five years are transients, and those present in only one year are occasionals or accidentals. Of the 36 resident species, 21 (58%) are soft substrate species and similarly, of the 24 transient species, 14 (58%) prefer sand and/or mud. Only nine (47%) of the 19 occasional species prefer soft substrate. The predominance of soft substrate species reflects the design of the harbor, which has almost no subtidal hard substrate habitable by fishes except at the entrance breakwaters.

Most of the soft substrate fishes of Marina del Rey are common to those observed at King Harbor and in shallow waters of the Santa Monica Bay and the Southern California coast [10,11]. Of the 12 species found at Marina del Rey that were not found at King Harbor (Table 1), almost all are soft substrate forms and most are shallow water wetlands species that are found only at the shallow sand area of Basin D. No such habitat exists at King Harbor, or, for that matter, elsewhere in the area with the exception of the little degraded wetland accessible at Ballona Creek.

Table 1

Fishes taken at Marina del Rey not recorded as adults in field samples from King Harbor Marina

Acanthogobius flavimanus (yellowfin goby)	s	w
Albula vulpes (Bone fish)	s	w
Anchoa compressa (deepbody anchovy)	s	w
Chitonotus pugetensis (roughback sculpin)	s	
Fundulus parvipinnis (Californica killifish)	s	w
Lepidogobius lepidus (bay goby)	s	w
Leptocottus armatus (staghorn sculpin)	s	w
Mustelus henlei (brown smoothhound)	s	w
Rimicola muscarum (kelp klingfish)	k	
Strongylura exilis (California needlefish)	s	w
Typhlogobius californiensis (blind goby)	s	
Umbrina roncador (yellowfin croaker)	s	

s = soft substrate inhabitant w = wetlands k = kelp

Comparison of wetlands species that have been recorded recently from wetlands and embayments in Southern California with those from Marina del Rey is presented in Table 2 [12-18]. Notice that Marina del Rey has the largest species list, but it has been sampled with more regularity than other habitats. The only obligate estuarine species, *Eucyclogobius newberryi*, was not taken at any of these bays and wetlands. Several species, i.e., *Paralichthys californicus* and *Cymatogaster aggregata*, utilize this habitat primarily as a nursery, while others show seasonality in their presence. All of the 17 principal species (Table 2) regularly occur in Marina del Rey. Several freshwater species found at Colorado Lagoon, Upper Newport Bay and Anaheim Bay were omitted from this comparison since Marina del Rey has no freshwater habitat.

SPECIES OCCURRENCES

The occurrence data on all species recorded from the harbor since 1977 are presented in Table 3. The number of species has varied considerably with season as well as by year. However the means, 41 species in the May and 39.9 in October, are not significantly different (Table 4). During 1987-88 the October samples exceeded those of May. When these data are reexamined by sampling method, the diver transects show considerable variation, due primarily to visibility at time of sampling. Visibility in Marina del Rey is usually poor. With less than five meter visibility, divers cannot estimate numbers and have simply recorded species. This is the usual type of data available, but occasionally, as in May 1988 and 1990, visibility was poor enough to considerably reduce the species tally. Similarly, with exceptional visibility the number of species is always high. It is possible that other sampling methods are also affected by visibility, most likely in the opposite direction. Gill nets and otter trawls both may be avoided under clear water conditions but the water at Marina del Rey rarely reaches that clarity.

Table 2

Fishes of shallow embayments and wetlands*

(COLAG = Colorado Lagoon [12]; BOLCH = Bolsa Chica [13]; SAR = Santa Ana River [13]; ALAB = Alamitos Bay [14]; HH = Huntington Harbour [14]; MDR = Marina del Rey [15]; BW = Ballona Wetlands [16]; UNB = Upper Newport Bay [17]; AB = Anaheim Bay [18].

	** COLAG	BOLCH	SAR	ALAB	HH	MDR	BW	** UNB	** AB
[1]*Acanthogobius flavimanus*		X	X	X	X	X	X	X	
Albula vulpes						X		X	
[1]*Anchoa compressa*	X	X	X	X	X	X		X	X
[1]*Anchoa delicatissima*	X					X		X	
[1]*Atherinops affinis*	X	X	X	X	X	X	X	X	X
Atherinopsis californiensis						X	X	X	
Citharichthys stigmaeus				X	X	X			X
[1]*Clevelandia ios*	X	X	X	X	X	X	X	X	X
[1]*Cymatogaster aggregata*	X	X	X	X	X	X	X	X	X
[1]*Embiotoca jacksoni*	X	X		X	X	X	X	X	X
Engraulis mordax	X	X	X	X	X	X		X	X
[1]*Fundulus parvipinnis*	X	X	X			X	X	X	X
Gambusia affinis							X	X	
Genyonemus lineatus	X			X	X	X	X		X
Gibbonsia elegans		X		X		X			
[1]*Gillichthys mirabilis*		X	X			X	X	X	X
Heterostichus rostratus		X		X	X	X	X	X	
Hypoglossina stomata				X		X			
[1]*Hypsoblennius gentilis*		X	X	X	X	X	X		X
Hypsoblennius gilberti				X		X			
Hypsoblennius jenkinsi				X	X	X			
[1]*Hypsopsetta guttulata*	X	X	X	X	X	X	X	X	X
[1]*Ilypnus gilberti*						X	X	X	
Lepidogobius lepidus		X		X	X	X			
Leptocottus armatus	X	X	X			X	X	X	X
Leuresthes tenuis	X						X	X	
Menticirrhus undulatus	X					X		X	X
[1]*Mugil cephalus*	X					X	X	X	X
Mustelus henlei		X				X		X	X
Myliobatis californica	X					X	X	X	
Paraclinus integripinnis				X		X			
[1]*Paralabrax maculatofasciatus*	X	X				X		X	X
Paralabrax nebulifer		X	X	X	X	X		X	X
[1]*Paralichthys californicus*		X	X	X	X	X	X	X	X
Phanerodon furcatus	X			X	X	X		X	X
Pleuronichthys ritteri				X	X	X		X	
Porichthys myriaster				X	X			X	X
[1]*Quietula y-cauda*	X					X	X	X	X
Rhacochilus vacca	X					X			X
Roncador stearnsi	X								X
Sardinops sagax						X			
Seriphus politus	X	X		X		X	X	X	X
Sphyraena argentea				X	X	X		X	
Strongylura exilis						X			
Symphurus atricauda				X	X	X		X	X
Syngnathus leptorhynchus		X	X	X	X	X		X	X
Synodus lucioceps						X			
[1]*Urolophus halleri*	X	X		X	X	X		X	X
Xystreurys liolepis				X		X			

*Larvae and adults only, fishes reported only from egg stages excluded. Marine harbors such as King Harbor and Outer Los Angeles-Long Beach Harbors are also excluded along with an occasional typically marine species noted rarely in these embayments.

**Several freshwater species (besides the salinity tolerant *Gambusia*) were recorded but not listed: *Dorosoma petensis*, *Lepomis cyanellus*, *Lepomis macrochirus*, *Morone saxatilis*, and *Pimephales promelas*.

[1]Principal species

Incidence of Fish Species in Marina del Rey, 1977-1990

Species	1977-79	1980	May 84	Oct 84	Oct 85	May 86	Oct 86	May 87	Oct 87	May 88	Oct 88	May 89	Oct 89	May 90	Oct 90	May 91
Acanthogobius flavimanus		x					x		x	x	x	x				
Albula vulpes			x							x						
Anchoa compressa			x		x	x		x			x					
Anchoa delicatissima									x							
Anisotremus davidsoni			x	x	x	x	x	x				x	x		x	
Atherinidae																x
Atherinops affinis	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Atherinopsis californiensis	x					x	x	x	x	x		x				
Atractoscion nobilis			x			x	x	x								
Cheilotrema saturnum		x		x	x	x	x	x	x	x	x	x	x	x		x
Chitonotus pugetensis											x					
Chromis punctipinnis				x	x	x	x	x	x	x	x		x		x	x
Citharichthys stigmaeus		x	x	x				x					x			
Clevelandia ios	x	x	x			x	x	x		x			x		x	
Clinidae Type A												x		x		
Clinocottus analis	x	x	x		x		x			x	x	x				x
Clupeiformes			x							x						
Coryphopterus nicholsii								x				x				
Cottidae		x														
Cymatogaster aggregata	x					x		x				x	x	x	x	x
Embiotoca jacksoni	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Engraulis mordax	x	x		x		x	x	x		x	x		x	x	x	x
Engraulididae			x			x										
Fundulus parvipinnis	x		x	x	x		x	x		x	x		x	x	x	x
Genyonemus lineatus	x	x	x		x	x		x				x		x		x
Gibbonsia elegans	x		x		x		x		x		x	x	x		x	x
Gillichthys mirabilis		x					x									x
Girella nigricans	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Gobiesox rhessodon		x			x	x		x				x		x	x	x
Gobiidae A/C		x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Gobiidae D			x	x	x				x		x					x

Table 3 (continued)

Incidence of Fish Species in Marina del Rey, 1977-1990

Species	1977-79	1980	May 84	Oct 84	Oct 85	May 86	Oct 86	May 87	Oct 87	May 88	Oct 88	May 89	Oct 89	May 90	Oct 90	May 91
Halichoeres semicinctus			x	x	x	x	x	x	x	x	x	x	x		x	x
Hermosilla azurea				x	x		x				x	x				
Heterodontus francisci			x	x								x		x		x
Heterostichus rostratus		x	x		x	x	x	x	x	x	x	x	x	x	x	x
Hippoglossina stomata		x				x										
Hyperprosopon argenteum				x		x										
Hypsoblennius sp.*	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
Hypsoblennius gentilis			x		x	x	x		x							
Hypsoblennius gilberti		x					x		x							
Hypsoblennius jenkinsi			x	x	x		x	x	x	x		x				
Hypsopsetta guttulata	x	x	x	x			x	x	x		x	x	x	x	x	x
Hypsurus caryi																x
Hypsypops rubicundus	x		x	x	x		x	x	x	x	x	x		x	x	x
Ilypnus gilberti			x	x		x	x	x		x				x		x
Lepidogobius lepidus						x										
Leptocottus armatus	x					x	x	x		x		x	x	x		x
Lythrypnus dalli		x														
Medialuna californiensis									x							
Menticirrhus undulatus						x	x		x	x	x	x				x
Micrometrus minimus			x	x	x	x	x	x	x	x	x	x	x	x	x	x
Mugil cephalus			x	x	x	x	x	x	x	x	x	x	x	x	x	
Mustelus henlei	x								x							
Myliobatis californica	x					x	x	x	x		x	x	x	x		x
Neoclinus stephensae		x														
Oligo/Clinocottus type A													x			
Oxyjulis californica	x		x	x	x	x		x	x		x	x	x		x	x
Oxylebius pictus														x		
Paraclinus integripinnis		x			x		x					x		x	x	
Paralabrax clathratus	x		x	x	x	x	x	x	x	x	x	x	x	x	x	x
P. maculatofasciatus	x		x	x	x	x	x	x	x			x	x	x	x	x

* = a mix of H. Gentilis and H. jenkinsi larvae too young to identify separately

Table 3 (continued)

Incidence of Fish Species in Marina del Rey, 1977-1990

Species	1977-79	1980	May 84	Oct 84	Oct 85	May 86	Oct 86	May 87	Oct 87	May 88	Oct 88	May 89	Oct 89	May 90	Oct 90	May 91
Paralichthys californicus	x	x	x	x	x	x	x		x	x	x	x	x	x	x	x
Phanerodon furcatus	x					x			x		x	x		x		x
Pleuronichthys coenosus					x				x							
Pleuronichthys ritteri			x		x		x	x	x	x	x	x	x		x	x
Pleuronichthys verticalis		x				x			x					x		x
Quietula y-cauda			x			x	x	x		x		x	x	x		x
Rhacochilus toxotes		x		x					x		x			x	x	
Rhacochilus vacca			x	x	x	x	x	x	x	x	x	x	x	x	x	x
Rhinobatis productus			x													
Rimicola muscarum	x		x	x												
Sarda chiliensis	x		x	x			x		x							
Sardinops sagax		x	x	x	x	x	x	x	x	x		x			x	x
Sciaenidae complx 2											x			x		
Scorpaena guttata						x						x				
Scorpaenichthys marmoratus			x									x	x			x
Sebastes auriculatus			x													
S. serranoides	x		x		x	x			x	x	x		x			
Semicossyphus pulcher = Pimelometopon pulchrum									x		x					
Seriphus politus	x	x	x	x	x	x		x	x	x	x	x	x	x	x	x
Sphyraena argentea	x			x	x	x			x	x				x	x	
Squatina californica					x											
Stenobrachius leucopsarus		x				x									x	
Strongylura exilis			x	x	x	x		x			x	x	x	x	x	
Symphurus atricauda	x	x		x												
Synodus lucioceps		x									x					
Syngnathus sp.	x			x	x									x		x
Syngnathus leptorhynchus			x					x	x			x			x	
Typhlogobius californiensis		x	x			x								x		
Umbrina roncador			x	x	x	x	x	x	x		x					
Urolophus halleri	x	x	x				x	x		x		x				x
Xenistius californiensis				x					x				x		x	
Xystreurys liolepis		x	x								x					

Table 4

Summary of number of species by month and year

	1977 1979*	1980 1981+	May 84	Oct 84	Oct 85	May 86	Oct 86	May 87	Oct 87	May 88	Oct 88	May 89	Oct 89	May 90	Oct 90
Diver Transects	14	-	20	24	19	22	20	18	24	15	24	22	22	13	20
Beach Seine	2	-	7	5	5	10	17	10	8	8	5	5	7	8	7
Gill Net	3	-	6	4	4	9	6	6	8	6	2	1	2	2	1
Otter Trawl	13	7	14	4	3	6	4	7	8	9	10	10	8	12	7
Ichthyoplankton (larvae only)	-	30+	10	7	9	11	7	9	10	8	4	4	3	14	11
Drop Net	-	6	-	-	-	-	-	-	-	-	-	-	-	-	-
Cryptic Fish	-	4	-	-	-	-	-	-	-	-	-	-	-	-	-
Creel Census or visual sighting	4	-	-	-	-	-	-	-	-	-	-	-	-	-	-
TOTAL NO SPECIES	35	44	47	37	38	46	41	39	46	33	39	43	35	38	35

* Quarterly sampling for Jan. 1977 through June 1978 and 1979.
+ Quarterly sampling for 1980: Aug. 1980, Sept. 1980, Jan 1981 and April 1981 (ichthyoplankton, drop nets, quinaldine stations, otter trawls)

Table 5

Summary of number of individuals by month and year

	Jun 77	Oct 77	Jun 78	May 84	Oct 84	Oct 85	May 86	Oct 86	May 87	Oct 87	May 88	Oct 88	May 89	Oct 89	May 90	Oct 90
Beach Seine	---	---	---	186	303	241	476	400	791	70	14135	486	1253	554	3550	750
Gill Net	---	---	---	80	19	17	42	13	56	27	65	49	20	5	263	1
Otter Trawl	618	212	238	136	14	6	17	13	26	18	95	251	34	59	33	209
TOTAL FISH				402	336	264	535	426	873	115	14295	786	1307	618	3846	954
TOTAL ICHTHYOPLANKTON				7567	2677	1767	2156	1550	6592	2255	2139	925	2353	286	2559	3524

GILL NET AND BEACH SEINE RESULTS

The gill net samples averaged 6.1 species (1.7 = SD) for the eight sampling periods from May 1984 - May 1988 while they have averaged only 1.6 (0.5 = SD) in the five sampling periods since May 1988. This is a significant difference and is difficult to explain. The beach seine data have been rather consistent except for an unusual collection on in the fall of 1986 17 species, and have averaged the same numbers as the trawl data, though there is no correlation between the samples.

The reduction in species in 1988-89 was largely due to the absence of gobies, generally annual species. Initially this was coincident with a bubble net placement between Basin D and the marina in an attempt to reduce coliform counts on the swimming beach. Such a net very likely affected recruitment of those annual young to the sampling site. It is also apparent, however, from the ichthyoplankton data that numbers of larval species were very low during 1988- 89. It may also be that the subsequent absence of larvae was due to introduction of PCBs into the marina sometime between October 1988 and October 1989 [8] that could have resulted in high larval mortality, as will be discussed further.

By contrast, if we look at the numbers of fish for the sampling where data applies (Table 5), the results are somewhat different. The gill net results show that the May 1990 catch was the highest since the origin of the study and was about three times the previous high which was recorded in 1984. However, these data represent only a large catch of topsmelt, much better represented in the beach seine data and not indicative of improvement in the marina. The lowest gill net catch, one individual, occurred in October of 1990, the next sampling period. The gill net data do agree with the beach seine data for 1990, in that a large number of topsmelt were taken in each during May; in fact the number taken in the beach seine ranked second to an exceptional catch in May 1988. Unfortunately, the total abundance data for the harbor always reflect the variability of this species. When topsmelt are removed from the data, little variation in numbers is recorded.

OTTER TRAWL AND ICHTHYOPLANKTON RESULTS

Otter trawl data show no true trends, but the highest number of species (14) was recorded in 1984 with the lowest in 1985 (Table 4). Recent catches have been well above the mean (7.9, 3.4 = SD). The otter trawl data generally do not show an abundance of fishes. The two samples with high counts represent primarily large numbers of young of year queenfish (*Seriphus politus*), though in 1984 there was a large number of juvenile barred sand bass (*Paralabrax nebulifer*) also taken (Table 5).

Since Since 1984 our ichthyoplankton sampling has collected 26 identifiable species as well as several unknowns (Table 6). One species, *Hypsoblennius jenkinsi* (= *Hypsoblennius* sp.) and Goby A/C (= a complex of *Quietula*, *Ilypnus* and *Clevelandia*) were present all seven years while *Sardinops sagax* and *Seriphus politus* were present in six of the seven. Six species occur only once, including *Oxylebius*, a cold water form. The single annual identification of *Ilypnus* reflects the only year larvae were large enough to separate from other members of the Goby A/C complex.

With the exception of three collections which averaged only 3.6 species (October 1988, May and October 1989), larval samples averaged 9.4 species per collection as compared to 8.0 species (3.2 = SD) for all periods. These data compare to annual means from King Harbor of between 6 to 17 species [12]. The highest number of Marina del Rey species, 14, was recorded in May 1990 [8]. During the quarterly samples of 1980, 31 different species were recorded, considerably more than in our biannual samples but about the same as our accumulated species list. A major difference since 1980 is the lack of six of the seven species of cottids but these are probably winter spawners and thus are not sampled in the May and October surveys.

The abundance data from the ichthyoplankton larval collections are quite variable (Table 7). There is a strongly seasonal aspect to this abundance, (Figure 3), with significantly more larvae present in May than in October.

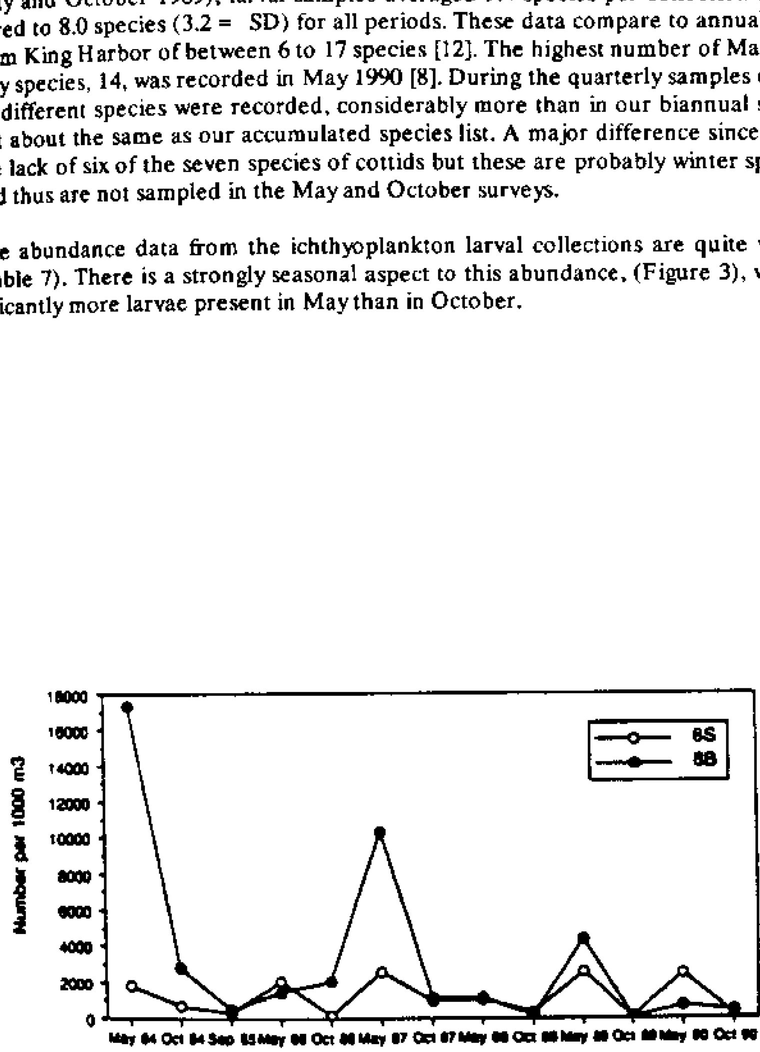

Figure 3. Seasonal Abundance of Ichthyoplankton from Station B.

Table 6

Annual variation in presence (P) of Ichthyoplankton in Marina del Rey

Species	1984	1985	1986	1987	1988	1989	1990
Hypsoblennius sp	P	P	P	P	P	P	P
Gobiidae sp A/C	P	P	P	P	P	P	P
Gobiidae sp D	P	P		P	P		
Engraulis mordax	P		P	P	P		
Engraulididae	P						
Genyonemus lineatus	P	P	P	P			
Clupeiformes	P				P		
Typhlogobius californiensis	P		P				P
Seriphus politus	P		P	P	P	P	P
Syngnathus leptorhynchus	P					P	P
Sardinops sagax caeruleus	P	P	P	P	P	P	
Symphurus atricauda	P						
Paraclinus integripinnis		P	P			P	P
Clinidae Type A		P		P		P	P
Gobiesox rhessodon		P	P	P		P	P
Pleuronichthys verticalis			P	P			P
Pleuronichthys ritteri		P		P			P
Paralichthys californicus		P	P				P
Hypsopsetta guttulata			P	P			P
Heterostichus rostratus				P			
Semicossyphus pulcher				P			
Hypsypops rubicundus				P	P	P	P
Cheilotrema saturnum					P	P	P
Anisotremus davidsonii						P	
Oylebius pictus							P
Oligocottus/Clinocottus Type A						P	
Ilypnus gilberti				P			
Stenobrachius leucopsarus			P				P
Atherinops affinis			P	P	P	P	P
Sciaenidae Complex II			P	P	P		P
Unknown			P	P	P	P	P

Table 7

Larval abundance by station and season (N/1000 m^3)

Surface Stations

	2S	5S	8S	Mean	SD
May-84	2023.9	2961.5	1803.5	2263.0	614.9
Oct-84	559.2	6794.5	686.4	2680.0	3563.8
Sep-85	697.4	3243.4	302.5	1414.4	1596.2
May-86	247.8	3774.4	2076.5	2032.9	1763.7
Oct-86	60.3	2115.3	203.8	793.1	1147.3
May-87	977.5	2551.0	2532.5	2020.4	903.2
Oct-87	377.0	2837.5	963.5	1392.7	1286.2
May-88	1155.5	925.3	1059.2	1046.7	115.6
Oct-88	2596.9	343.0	317.4	1085.8	1308.8
May-89	620.3	5358.1	2574.0	2850.8	2381.0
Oct-89	287.4	678.2	121.8	362.5	285.7
May-90	686.3	185.5	2492.6	1121.5	1213.5
Oct-90	968.9	1898.9	148.4	1005.4	875.8

Bottom Stations

	2B	5B	8B	Mean	SD
May-84	3452.9	17899.9	17322.0	12891.6	8179.3
Oct-84	1342.6	3875.5	2806.7	2675.0	1271.5
Sep-85	3446.2	2437.4	481.3	2121.6	1507.4
May-86	2090.3	3261.7	1490.7	2280.9	900.8
Oct-86	665.6	4200.7	2054.9	2307.0	1781.0
May-87	5224.2	18002.6	10267.6	11164.8	6436.3
Oct-87	2939.0	5307.8	1110.3	3119.0	2104.5
May-88	2767.3	5776.0	1153.7	3232.3	2346.0
Oct-88	1776.2	344.9	177.4	766.2	878.7
May-89	702.9	511.5	4353.6	1856.0	2165.2
Oct-89	272.1	279.8	75.1	209.0	116.1
May-90	3753.3	7538.1	702.8	3998.1	3424.2
Oct-90	528.2	11151.6	389.5	4023.2	6174.0

This is especially the case for the most inland site, Station 8. The fewest larvae were taken during October 1989 ($\bar{x}$ = 285.7, N = 6) which can be compared to maximum means of 7515.3 in May 1 1984 and 6,593.6 in May 1987. The mean for all May samples was 3408.0, while in October was 1401.4. This difference is highly significant (t = 2.915, p = .005). The mean abundance of larvae in Marina del Rey is very similar to that recorded at King Harbor.

FACTORS INFLUENCING THE ICHTHYOFAUNA

The normal oceanographic regime influencing Santa Monica Bay was summarized earlier. However a major influence on the fishes of the bight is the semiregular oceanographic occurrence, the El Niño-Southern Oscillation. Two of these events occurred since 1973; one in 1977-78 was a minor event from the standpoint of its worldwide effect while the second, in 1982-84, was a major environmental occurrence. Interestingly, in Santa Monica Bay, the first of these had the most dramatic effect on fish populations. During the early 1970's, a cold water anomaly had allowed the development of a fauna with a strong, cold temperate element, especially the flatfishes and rockfishes. This element disappeared rapidly with the onset of the 1977-78 El Niño and, in fact, has not returned. Dover and Rex sole were abundant in the deeper water of the bays, speckled sanddabs were common in shallow water, while blue rockfish dominated King Harbor and stripetailed and halfbanded rockfish were common in trawl catches. With the disappearance of many of these species, the average catch of soft bottomed fishes in trawls declined rapidly [19]. This change has not been reversed by recent events. In fact, the great El Niño of 1982-84 exacerbated the problem and stimulated the movement of a number of species commonly found to the south into Santa Monica Bay.

The earliest fish studies of Marina del Rey, conducted in 1977, were just at the time of onset of the little El Niño. During those quarterly surveys, 31 species were recorded of which 10 were wetlands-back bay species. The remainder showed no evidence of the cold water anomaly that was ending, with the exception of croakers which typify our shallow sandy environment but tend to move off shore during warming events (white croakers, for example, prefer about 11.5°C water). It is possible that the shallow conditions of the marina which cause a greater range in temperature do not allow cold temperate species to persist there even during cold anomalies.

The 1982-83 period of the great El Niño was not sampled at Marina del Rey but data from King Harbor demonstrated a significant decrease in both abundance and diversity of species for those years. The 1984 marina samples, the first of our standardized sampling, were taken at the end of this event, although very high water temperatures persisted in the Southern California Bight through October 1984 [3]. Even though some different techniques were used in pre-1984 samples, 31 to 32 species were recorded in samples taken quarterly. During 1984 we recorded 60 species in biannual samples.

Comparing these samples, 21 species found in 1984 were not seen in the previous surveys: 10 of these are rocky wall species that were not sampled (except for cryptic species) during the earlier period, but 9 of the 21 are warm preferring species typically favored by El Niño events. Of the 17 species found in the period between 1977-80 but not found in 1984, four were rocky wall species and five were forms that tended to disappear during warm water conditions. The important warm water back bay species, such as the bone fish *Albula vulpes* and the striped mullet *Mugil cephalus* may have arrived during those periods. *Albula* occurs in much higher abundance in Marina del

Rey and the Venice Canals than in any other bay or estuarine environment in the Southern California Bight, a unique circumstance (L.G. Allen, pers. comm.).

Unfortunately for our observations of natural variation, warm conditions still exist, though they have cooled considerably since 1984. No important faunal shifts have been observed since 1984 although several cooler water species have occasionally been taken, for example, the horny head turbot *Pleuronichthys verticalis* and the white surfperch *Phanerodon furcatus*.

URBAN IMPACTS

A number of actual and potential pollution problems exist in Marina del Rey as in all urban marinas. Trash dumped into Ballona Creek finds its way into the marina on tidal changes, accidental spills may enter through storm drains and flood control channels, and streets and storm drains flush oil, grease and toxic materials into the marina during heavy rainfall. Levels of heavy metals, pesticides and chlorinated hydrocarbons as measured in marina sediments are excessive [2-8] but do not correlate with observed measures of infaunal biodiversity and abundance. The infauna of the harbor is predominantly polychaeta worms, while nematodes and oligochaetes sometimes occur in very large numbers at a disturbed station. Pollution-sensitive organisms such as crustacea, echinoderms and molluscs are reduced in numbers.

Infaunal decreases in recent years may have been associated with several factors. Levels of the antifouling compound tributyltin used in boat paints were sufficient to cause chronic reproductive inhibition in representative species tested [6]. Tributyltin, one of the most toxic substances ever introduced into marine waters, has been banned on most vessel hulls since 1988 and levels have decreased greatly since then. Relatively high levels of Chlordane and DDTs, all of which are banned, occur in marine sediments, largely contributed by the flood control channels and storm drains, although illegal dumping may occur.

PCBs (Aroclors) used in electrical systems pumps, compressors, hydraulic fluids, printing inks and as plasticers, have been banned for many years. It was thought that their occurrence had ceased in the marina between the 1970's surveys and the 1980's, although analytical detection limits were greatly improved. PCBs had not been identified as present in marina sediments until the October 1989 survey, and the introduction of Aroclor 1254 and the relatively uncommon Aroclor 1260 were sufficient to institute a resampling of sediments as well as examination of fish body burdens for protection of public health.

The sources of the PCBs and also increased DDTs are unknown but excavations on former industrial sites near the marina may have resulted in increased contaminated runoff through flood control channels or storm drains. No dumping has been reported. The area was heavily industrial during World War II, with no controls on dumping or abandoning wastes that may have been graded into the site.

FISH TISSUE POLLUTANT BURDEN

In January - February 1990, 32 fish specimens representing 12 species were analyzed for body burdens of PCB 1260, DDD, DDE and DDT in muscle, with liver and gonad analyzed in three specimens [8]. Ranges are shown in Table 8 which follows.

Table 8

Range of Chlorinated Hydrocarbons in (ug/kg wet wt:ppb)

	PCB (Arodor 1260)	DDD	DDE	DDT
muscle	15-298	Ud-102	Ud-1,665	Ud-60
gonad/liver	548-4,27	127-311	525-6,806	79-17

(ud = undetected)

The muscle levels found are considered safe for normal human consumption, whereas liver/gonad tissue would be suspect. *Myliobatis californica* (batray) liver/gonad had the lowest level and *Synodus leucoceps* (California lizard fish) had the highest, with *Pleuronichthys ritteri* (spotted turbot) intermediate. The same pattern was true for DDD, DDE and DDT, illustrating the difference in sequestering by different individuals or species. The higher levels of chlorinated hydrocarbons are capable of producing reproductive inhibition in *Genyonemus lineatus* (white croaker) and other species [21,22]. There is wide variation in the uptake and sequestering of chlorinated hydrocarbons [23]. There is also seasonal variability as to the levels in liver and gonads, with large amounts leaving the body in reproduction [24]. Thus eggs would be heavily contaminated, inhibiting egg and larval development.

In spite of the presence of contaminants in sediments, some areas of the marina have a high level of production of soft bottom invertebrate fauna attractive to a number of fish species.

Efforts to correlate the incidence of existing invertebrate species with any of the contaminants, trace metals and pesticides have not been productive. This is probably due to the virtual absence of sensitive species of molluscs, crustaceans and echinoderms.

THE MARINA AS A WETLAND HABITAT

Ballona wetlands once covered an area from Venice on the north to the bluffs on the south and inland almost to the present San Diego Freeway. Ballona Creek, which occasionally was, in the past, also the bed of the Los Angeles River, drains much of the west central Los Angeles Basin. All that remains of the natural wetlands is the degraded area on the south side of Ballona Creek Flood Control Channel, an area that has been largely shut off from tidal flushing for many years following concretizing the local rivers in the 1920s and 1930s.

The marina has thus acted as an important refuge for wetlands fish species, as has been indicated in the foregoing discussions. Some ninety percent of the natural, sheltered, shallow water habitats in the Los Angeles region have been lost [25], making the protection of the marina environment of great importance.

Plans for restoration of the degraded Ballona wetlands south of Ballona Creek and for a new marina basin on the north side offer great opportunities to link the existing marina and the wetlands into a single hydrological unit [26,27]. This would offer the wetlands fish species, as well as the larval and juvenile stages of coastal fishes, a much greater range of habitats — food, temperature, salinity and substrates — than presently exist, as illustrated in Figure 4 [26], a conceptual scheme that offers great promise.

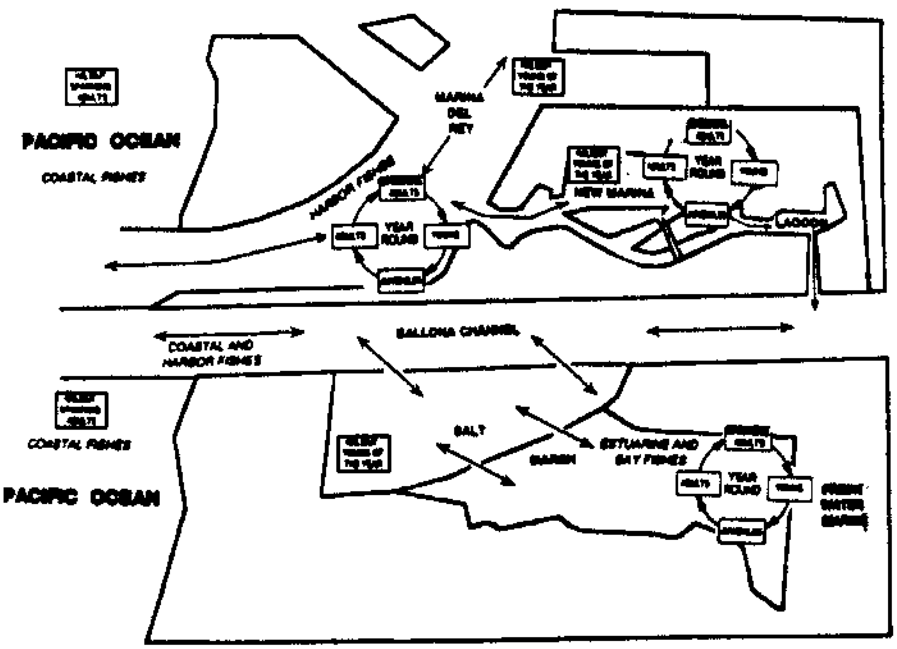

Figure 4. Conceptual relationships linking existing Marina del Rey and the proposed marina with Ballona Wetlands (after Davis, 1991 [26]).

CONCLUSIONS

Harbor and marina environments can provide important habitat for coastal or wetlands species. If wetlands species are to be protected and enhanced, they must be supplied with an adequate shallow water habitat, which is not necessarily compatible for the primary purpose of a harbor/marina facility. The swimming beach at Marina del Rey is an exception that provides such a habitat. Further, in order for a marina to support a variety of fishes, there should be as much available sloping rocky substrate in the subtidal area as possible. In this respect, Marina del Rey falls short because there is little rocky substrate except at the opening breakwaters, with most walls constructed as vertical concrete L- walls. It is possible to design rocky groins in a marina that are compatible with vessel accommodation and still provide desirable fish habitats.

Harbors and marinas must be designed with adequate circulation since they are, per se, habitats. Inland harbors like Marina del Rey and Huntington Harbour have poor tidal circulation. In the future, all design of new harbors should be required to use circulation models that favor the needs of the biological communities that will certainly inhabit them as well as the needs of vessel users. Harbors not only supply homes for resident and transient fishes and other marine organisms but their quiet waters often serve as an important nursery area for many deeper water marine species. Although the purpose of a marina or harbor is a boating-shipping facility, it is always an "attractive nuisance" to fishes, especially as a nursery; it must, therefore, function as a fish habitat. New facilities must be designed to fulfill this function, and where possible, existing facilities augmented to improve the plight of scarce wetlands species.

The continued expansion by crowding of new floating docks into formerly open water only exacerbates the already low circulation, although it may provide habitat for more fouling communities and provide shelter for small fish.

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Invasive Exotic Plants: Threats to Coastal Ecosystems

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Abstract. Exotic plants that invade coastal salt marsh habitats can reduce mudflat area and/or displace native plants. Several situations promote their invasion, including enhanced dispersal, substrate disturbance, hydrologic modifications (especially reduced soil salinity), and persistent changes in the environment. Case studies indicate the type of problems that follow exotic species invasions in California salt marshes. A common feature is that short-term alterations in habitat initiate long-term problems that are not easily solved. Management strategies should therefore focus on prevention.

INTRODUCTION

Invasive exotic plants threaten the morphology and composition of coastal salt marshes, both by invading adjacent mudflats and displacing native vegetation. These problems have long been recognized in Great Britain, and have developed more recently in North America. Examples where detailed information is available indicate the extent and severity of management problems, the kinds of disturbances that promote invasions, and the effectiveness of various control measures. These examples are reviewed herein, along with newer case studies from California.

Many exotic species have travelled the world's oceans and invaded foreign coastlines. Those that have done so at the hand of man, or followed the migration of man to new continents, are generally termed exotic. While it may not seem fair to distinguish species carried by man from those that arrived on the feet or in the guts of other well-known dispersal agents (notably birds), it emphasizes the fact that European man has had an inordinate influence on the globe, such that seeds carried on his soles have a much higher probability of becoming established on foreign soil that has been cleared, plowed, or otherwise modified.

Unfortunately, records of the arrival and early spread of exotics are rare, and some man-introduced species are accepted as natives. When aliens become nuisances, it is useful to know the time of arrival and conditions under which they began to spread. This can help in predicting future spread and in identifying control measures. Indirect evidence can be obtained from pollen profiles in wetland sediments, which record the sequence of immigrant weeds in central and southern California (Mudie and Byrne 1980). Adobe bricks include seeds of weeds that were present at the time of their production. Herbarium collections and species maps may indicate times and species that were historically rare and places where they later became abundant.

Underlying this paper is a conservation ethic, that native species should dominate California's natural coastal wetlands, and that recently introduced species should be controlled, if not eliminated. The rationale behind this judgement is straightforward: 1) California's coastal wetlands are small and few—there are about 130 in the entire state. 2) The remaining wetlands have been highly modified and severely reduced in area—losses of 75-95% are commonly estimated. 3) The native plants are essential to many native animals (e.g., insects with high host specificity) and preferred by others—the native vegetation performs a variety of functions such as providing food, shelter, and nesting materials, that may or may not be replaceable by alien species. 4) Exotic species can spread rapidly and displace native plants—but the conditions that promote invasion cannot always be uncovered. 5) Once established, naturalized exotics are difficult, if not impossible, to eradicate.

California's coastal wetlands are a limited natural resource that is jeopardized by invasive exotic species. Because the risks are high, complacency is not an option. Rather, management goals should include: preventative measures where exotics have not yet invaded, eradication where invaders are not yet abundant, and active, continuing control of well-established exotics.

THE PROBLEMS IN COASTAL WETLANDS

Two types of management problems result from plant invasions in coastal wetlands—the elimination of open mudflat and the replacement of native vegetation.

1) The reduction of mudflat area occurs in intertidal wetlands where the native vegetation is unable to grow at elevations extending downslope toward mean sea level. A species with greater inundation tolerance may arrive and naturalize rapidly where there are no competitors to slow its establishment. In Willapa Bay, Washington, the invasion of *Spartina alterniflora* into mudflats threatens a major oyster industry. The invasion of intertidal flats in Great Britain by non-native cordgrasses (cf. Case study 1a) has reduced feeding habitat for native wading birds (Goss-Custard and Moser 1988). Cordgrass control is encouraged, although in some locations, e.g., England's Poole Harbour, the cordgrass is valued for its shoreline erosion control capability (Gray 1985). Even there, however, the managerial opinion is that "a little is good, a lot is not" (A. Gray, pers. comm.).

2) Where exotic plants invade vegetated areas, they may displace native species. In southern California, the invasion of the New Zealand mangrove, *Avicennia marina*, into native cordgrass (*Spartina foliosa*) habitat is viewed as detrimental to both plants and birds (cf. Case study 1d). The local escape of a few mangroves planted in Mission Bay, San Diego, prompted concern that the trees would displace cordgrass, which is the preferred nesting habitat and nesting material of the light-footed clapper rail (*Rallus longirostris levipes*), and that the trees would also provide roosting places for raptors, which readily prey on rail chicks (P. Jorgensen, Manager, Tijuana River National Estuarine Research Reserve, San Diego, pers. comm.). An active eradication program is underway, with annual visits to hand-pull seedlings (M. Pruitt-Jones, former Manager, Kendall-Frost Reserve, Univ. of California, San Diego, pers. comm.).

The indirect effects of shifting vegetation composition are not well known. While many wetland animals may lack species-specific requirements for plant canopies or nesting materials, insects are noted for their tight dependencies on single host species. Some are limited to individual families or genera of plants; others are restricted to single species or life history stages (e.g., flowers or seeds) of a host plant. I assume that these same dependencies are present among the insect inhabitants of coastal wetlands; yet wetland insect communities have received very little attention (C. Nagano, U.S. Fish and Wildlife Service, Endangered Species Office, Sacramento, pers. comm.) and only a few of the plant-insect linkages have been recorded. A rare butterfly, the wandering skipper (*Panoquina errans*) appears to be restricted to one salt marsh grass species (*Distichlis spicata*), while a scale insect (*Haliaspis spartina*) is specific to cord grass (*Spartina foliosa*).

Shifts in vegetation composition are not always viewed as detrimental. Recent and rapid vegetation changes are taking place along the Swan and Canning River estuaries near Perth, Western Australia (cf. Case study 2c). *Typha orientalis* is rapidly expanding into native salt marsh, where it replaces *Juncus krausii* and other native species (Pen 1983). Like the salt marshes of California, the western coast of Australia has few coastal wetlands, and the displacement of salt marsh vegetation is viewed as a management problem. However, because the native purple swamp hen (*Porphyrio porphyrio*) uses habitat dominated by *Typha* species, the managerial attitude is that some *Typha* expansion is all right (R. Atkins, Waterways Commission, Perth, pers. comm.).

Managers are generally more concerned with animal habitat losses than with shifts in vegetation composition per se. Threats to rare and endangered plant species and major changes in plant type are the exception. The salt marsh bird's beak (*Cordylanthus maritimus ssp. maritimus*) is an endangered plant species that occurs at the landward edge of intertidal salt marshes in southern California, where its habitat is frequently disturbed and weedy species threaten to invade, although no displacements have been documented. The goldfields (*Lasthenia glabrata*) is a rare plant in southern California (Ferren 1985). At one remnant population, Los Peñasquitos Lagoon, it occurs in close association with an exotic annual (*Cotula coronopifolia*; J. Boland, SDSU, pers. comm., and pers. obs.). Invasion of Florida's brackish wetlands by cajeput trees (*Melaleuca quinquinervia*, cf. Case study 1e) has prompted a costly control program aimed at retaining the marsh character of the vegetation. These two concerns merged when cajeput trees were deliberately introduced to Tijuana Estuary adjacent to habitat occupied by the salt marsh bird's beak (cf. Case study 1e). Tree removal was required after the invasive nature of the exotic was publicized and concerns for the native vegetation were voiced (pers. obs.). It was a costly error.

In at least one case, an exotic has become the overwhelming dominant of a major coastal wetland. The recent discovery (Spicher and Josselyn 1985) that the dominant cordgrass in Humboldt Bay is an exotic from Chile (*Spartina densiflora*) rather than the California native (*S. foliosa*) has prompted concern that this invader might displace the native cordgrass over a broader geographic area (M. Josselyn, Assoc. Professor of

Biology, San Francisco State Univ.; P. Kelley, Calif. Dept. of Fish and Game, Sacramento, pers. comm.).

WHAT ALLOWS OR PROMOTES INVASION BY EXOTICS?

The situations that allow invasion by exotic wetland plants can be identified by considering what prevents their establishment in our least disturbed wetlands.

1) Dispersal. If the physical, chemical, and biological attributes of the habitat are suitable for species that are not present, then the only limiting factor is the availability of propagules. Deliberate or accidental introduction of seeds/plants will then allow establishment (e.g., planting of mangroves in Mission Bay).

2) Disturbance. If the abiotic environment is suitable, but native inhabitants deter establishment (e.g., through shading of seedlings or consumption of seeds), then some disturbance to the canopy or substrate may allow exotics to establish. Removal of vegetation may simultaneously alter the substrate. For example, disking exposes mineral soil and creates suitable microsites for seed germination.

3) Temporary relief from environmental stress. If the abiotic environment is suitable for growth and survival, but too stressful (e.g., hypersaline) for seed germination and initial establishment, then a short-term reduction in stressful conditions (e.g., a prolonged flood) may permit invasion (e.g., *Typha domingensis* invasion and persistence in San Diego River Marsh; cf. Case study 2b).

4) Prolonged change in environment. If the abiotic environment is unsuitable for establishment and growth, then a persistent shift in environmental conditions may allow invasion (e.g., *Rumex crispus* invasions following prolonged impoundment of freshwater and chronic deposition of sediments from upstream erosion, at Los Peñasquitos Lagoon; cf. Case study 2d).

5) Combinations of the above. If two or more of the above limiting factors are removed, invasion moves from the possible to the probable. Such was no doubt the case for the early invaders whose seeds travelled to new lands in cargo ships, fell upon shores that were being developed into ports, moved up newly constructed navigation channels, or encountered estuaries where hydrology was being altered by changing land use patterns upstream in the watershed. Multiple factors appear to be responsible for the rapid advance of *Typha orientalis* into salt marshes of Western Australia, where street runoff from new housing developments is discharged through newly cut channels across the adjacent salt marsh. Neither lowered salinity nor substrate disturbance is sufficient to allow *Typha* seedling establishment, but the combination allows seedlings to establish in the drainage channels; thereafter, *Typha* invades the native marsh by vegetative expansion of adult plants (Zedler et al. 1990).

CASE STUDIES

A number of invasions in coastal wetlands have been described in recent years, and although the specific events that caused their establishment and spread are undocumented, their probable cause is inferred from their association with certain changes. Some have followed deliberate or accidental species introductions, some have followed major hydrologic changes, and others are common wherever substrates are disrupted. Each invasion may well have had multiple causes.

1) Invasions that followed deliberate or accidental species introduction

a) The classical story of plant species invasion in salt marshes is that of *Spartina townsendii*/*S. anglica* in England. In the 19th century, *S. alterniflora* appeared in Europe, where marsh grasses are valued for shoreline erosion control and marsh build ing through sediment accretion. It hybridized with the native *S. maritima* to form an infertile hybrid, *S. townsendii*, which was first noted in 1870. The hybrid later underwent chromosome doubling to form a very aggressive and fertile polyploid, which was noted in 1892 and later named *S. anglica*. With its increased vigor and the potential for seed dispersal, the new species was able to spread rapidly. In Poole Harbour, southern England, it was first recorded around 1880; by 1924 it had formed a marsh of more than 775 ha (1913 ac) and raised the topography by more than a meter in many areas. The plants at Poole Harbour then supplied seeds and propagules for a wide range of sites around Great Britain, the European continent, Australia, New Zealand, North America, and China (Gray 1985, Barnes 1977). On the U.S. Pacific Coast, it is currently known from Puget Sound, Washington, and San Francisco Bay, California (Spicher and Josselyn 1985).

The primary management concern about the ever-expanding distributionof *Spartina anglica* is the loss of mudflat habitat for shorebird feeding. Dense vegetation changes the character of the substrate and reduces habitat for the birds' preferred invertebrate prey. As Long and Mason (1983, p. 137) put it, "*Spartina* overgrows the mudflats and renders them useless for feeding by waders, who also dislike roosting in the tussocky growth." There is no lower-marsh bird species like the clapper rail that can take advantage of the grass. In addition, there is concern about the species' potential for outcompeting the native cordgrass, as well as dislike of exotic invaders by a populace that is well-informed about nature. A control program using herbicides is underway in northwestern England (Long and Mason 1983). In some areas, there have been major die-backs without management action, for which the causes are unclear (A. Gray, pers. comm.). For example, only about half of its previous maximum population persists in Poole Harbour (Gray 1985).

b) *Spartina densiflora* is a dominant in the intertidal salt marshes of Humboldt Bay. It is a caespitose (bunch) grass that has also begun to invade San Francisco Bay. Until recently, it was thought to be a ecotypic variant of the native *S. foliosa*, which is a turf-forming species. However, Spicher and Josselyn (1985) indicate that it is a Chilean species that was probably carried in ships that brought lumber from northern California to Chile and returned with ballast material collected from the Chilean shoreline. In San Francisco Bay, *S. densiflora* occurs slightly higher in the intertidal zone than S. foliosa and out competes the native *Salicornia virginica* (pickleweed; Josselyn and Buchholz 1984). Its spread in San Francisco Bay is of concern to botanists and managers, and Josselyn and Buchholz, in conjunction with Spicher (1984), devote a chapter of their Marsh Guide to the topic. The following summary is taken from their work in Marin County:

The species was introduced to Creekside Park in 1976 and expanded to a 14 km diameter range by 1984. It is also abundant on Corte Madera Creek and was planted for landscaping purposes in Greenwood Cove of Richardson Bay, where it is spreading from seed. Its seeds will germinate in sea water, and perhaps at higher salinities, and they are produced prolifically some two months earlier than those of *S. foliosa*. These characteristics and its high productivity make it a threat to the *S. virginica*-dominated marsh and the upper portion of the *S. foliosa* habitat. However, since the species appears to thrive in salinities lower than those tolerated by *S. virginica*, it is not expected to replace the upper portion of the native *S. virginica* marsh. While information on its impact on animal populations is lacking, these authors state that "Until the evidence supports that *S. densiflora* is not detrimental, all efforts should be made to control its spread to other locations in the bay."

c) *Spartina alterniflora* occurs in San Francisco Bay at the mouth of the Alameda Creek Flood Control Channel, as well as about 3 km to the south. According to Spicher and Josselyn (1985), the reason for its introduction and date of arrival are unknown. Aberle (1990) reports that *S. alterniflora* was introduced to Wallapa Bay, Washington, in the late 1800's as packing for oyster spat shipped from the East Coast. It was later planted in various areas of Puget Sound to stabilize shorelines and provide cover for waterfowl hunters. The species is now considered a major pest species in both San Francisco Bay and Puget Sound, and managers are seeking control methods (Aberle 1990).

d) *Avicennia marina*, a white mangrove from New Zealand, was planted in the University of California's Kendall-Frost Reserve at Mission Bay, to supply leaves for scientific study. From an initial planting, the species naturalized and the population threatened to change the character of the remnant salt marsh. Concerns that the trees would alter habitat for the endangered light-footed clapper rail and provide roosting sites for raptors that feed on rail chicks led to an eradication plan. In recent years, all visible individuals have been removed by hand in an annual effort to eliminate the species. It is likely that the species will eventually be controlled, although continual surveillance and removal are needed to deplete the seed bank.

e) *Melaleuca quinquinervia*, the cajeput tree, is native to wetlands in Australia. In southern Florida, this species has escaped from horticulture and is a major pest in freshwater wetlands. It is a prolific seeder that can store seeds on the tree in capsules (millions of seeds per tree) for several years without loss of viability. Release can occur simultaneously following a stimulus such as fire (Drew and Schomer 1984). This species was planted along the periphery of Tijuana Estuary by homeowners in Imperial Beach, San Diego County, who wanted to view trees along the marsh-sidewalk border. Its potential for invading the salt marsh was not considered during the planning process, because local managers were unaware of its pest status elsewhere. Although it is known to tolerate saturated soils, its salinity tolerance has not been tested. Thus, landscapers installed drip irrigation and planted the trees in imported soil. The trees established well but were later removed after the management concerns were identified. The tree is widely planted in San Diego, where evergreen trees that can grow in well-irrigated lawns are highly prized.

f) *Myopórum laetum* is an evergreen horticultural shrub that grows to small-tree size. Introduced from New Zealand, this species is a conspicuous but localized invader of the marsh periphery. It is abundant along the railroad that crosses Carpinteria Marsh and in the campground at Santa Clara Estuary. A few individuals are present near the Tijuana Estuary salt marsh. It appears to be quite salt tolerant but sensitive to inundation. Munz (1974) lists it as naturalized near Ventura.

g) *Carpobrotus edulis* (Hottentot fig, also called ice plant) has been widely planted along freeways, where it forms dense, monotypic mats of succulent leaves that resist drought, fire, and erosion. Its propensity for vegetative spread and ease of transplantation as short branches (it readily roots at the node) are the same characters that confer weediness. Eradication is likewise difficult, because remnants of plants regenerate the clones, and seeds germinate in the disturbed sites where adult plants have been pulled. At Los Peñasquitos Lagoon, San Diego County, the Dept. of Parks and Recreation has attempted a control program, which will only be successful if there is continual maintenance to remove new sprouts and seedlings (W. Tippets, Dept. of Parks and Recreation, San Diego, pers. comm.). This species has invaded a wide range of coastal habitats including strand and dunes (Williams and Williams 1984), chaparral (Zedler and Scheid 1988), and bluffs and salt marsh.

2) Invasions that followed hydrologic modifications

Both saline and fresh water wetland species are tolerant of inundation, and high soil moisture seems to be a requirement, at least for establishment. However, the salt marsh exists as a community more because its component plant species can tolerate high salinity than because of a basic physiological requirement for concentrated (3-4%) salt. Under lower salinities most species can grow better—but populations are low or absent in less saline habitats, because other species are likely to be more competitive. Species from fresh and brackish marshes are unable to invade salt marshes because seeds and/or seedlings are intolerant of higher salinity. Thus, changes in salinity brought about by prolonged reservoir discharge, irrigation runoff, inflows from street drains, or wastewater discharge can all shift species distributions downstream into the estuarine marshes. Even the native species that encroach on salt marsh habitats are considered aliens if their distributions expand under unnatural hydrologic conditions.

a) *Cotula coronopifolia* (brass buttons) is an herbaceous, succulent perennial from South Africa. It is common in both fresh and saline wetlands, including overwatered lawns. Its seeds germinate at 10 ppt salt but not 20 ppt (Zedler and Beare 1986). The species is widespread along the Pacific Coast. It is common in areas that accumulate winter rainfall, such as depressions within salt flats of the upper intertidal zone and in open mudflats that receive freshwater runoff. At Tijuana Estuary, it has invaded intertidal flats in areas downstream of Mexican sewage spills.

b) *Typha domingensis* (cattail) is widespread geographically but is not a salt marsh species. Its invasion of the San Diego River Marsh following the 1980 flood and prolonged period reservoir discharge led to a detailed study of factors that prevented its occurrence prior to 1980 and allowed its invasion during spring 1980 (Beare 1984, Beare and Zedler 1987), as well as a monitoring program to document its gradual decline (Zedler and Beare 1986). Experimental studies showed that very few seeds germinate below 20 ppt salt, and that seedlings required several months of low salinity soils in order to develop rhizomes and persist in tidally inundated soils. Field monitoring of soil salinities indicated that the San Diego River Marsh was brackish (in this case, below 10 ppt) for most of 1980. There was a strong positive correlation between streamflow and soil salinity, which supported the cause-effect relationship between reservoir discharge and altered marsh soil salinity. The abundance of cattails also tracked streamflow variations. Their expansion rate was greatest in1980, when seedlings established over most of the intertidal salt marsh. Biomass dropped in 1981 and 1982, which were low-flow years. During 1983, a year of very high streamflow, the cattail population flourished through vegetative regrowth (not seedling establishment), as soils were again brackish for a portion of the growing season. Since that time, the population has declined steadily, but it has not yet been eliminated from the marsh. In experimental conditions, Beare (1984) found that rhizomes could resprout even after being held at 4.5% salt (about 1.3 x seawater) for an entire year. Its continued presence, even in very low numbers, makes reinvasion likely, since plants can expand vegetatively at salinities higher than those required for seed germination. The normal "low-salinity gap" is too brief for cattail establishment, but a prolonged low-salinity gap allows invasion.

Elsewhere in California, this and other species of *Typha* are known to develop populations that spread into saline marshes. Examples are the marshes upstream of San Elijo Lagoon (D. Racine, Calif. Dept. of Fish and Game, San Diego, pers. comm.), pocket marshes adjacent to drains that enter Upper Newport Bay, and diked areas within south San Francisco Bay (J. Haltiner of P. Williams Assoc., San Francisco, pers. comm.). Although there are few quantitative records of such expansions and no studies to explain their advance, it is likely that the situation is similar to that for *T. domingensis* in the San Diego River Marsh. The records for *T. orientalis* in Australia (below) support the generality of the low-salinity gap phenomenon.

c) *Typha orientalis* is a native of Australia, but not of the Swan and Canning River estuaries near Perth, Western Australia. The situation is relevant to California wetlands, because the climate is similar, and rainfall is a winter event, followed by prolonged summer drought. *Typha orientalis* is rapidly invading and replacing native salt marsh in areas where street drains have increased runoff and reduced soil salinities (Pen 1983). Experimental studies (Zedler et al., in press) with seeds, seedlings, and rhizome-bearing adults show that salinity tolerance increases with age, and that vegetative growth can occur in conditions far too saline for seedling establishment. The seedlings are not good competitors, so that disturbed soils must coincide with low salinities in order for seedlings to survive. Thus, invasion is restricted to places where soils are disrupted and salinities are nearly fresh for unnaturally long periods. Street drains cut through the marsh sod afford exactly those conditions. Once seedlings develop rhizomes, the plant becomes more salt tolerant and more competitive . Thus, a single successful seedling can establish a clone and expand vegetatively into the native marsh.

c) ***Rumex crispus*** (Curly dock) invades the periphery of salt marshes if low salinities persist beyond the normal winter wet season. Germination tests (Zedler and Beare 1986) indicate that salinities below 10 ppt are necessary for recruitment from seed; it is a perennial but does not reproduce vegetatively. At Los Peñasquitos Lagoon, San Diego County, ***Rumex crispus*** has become a conspicuous component of the higher marsh, sharing dominance with *Distichlis spicata*, the native saltgrass. Its abundance has increased significantly in recent years, following prolonged periods of closure and inundation by local runoff (per. obs.).

In many of California's coastal wetlands, tidal flushing is discontinuous. This is especially true of smaller wetlands and of those with relatively small watersheds. Their ocean inlets tend to become closed when sand accumulates during summer. Any modifications, such as filling and sediment accretion, that reduce the lagoon's tidal prism may increase the frequency and duration of closure. If closure is followed by periods of low streamflow that are sufficient to reduce lagoon salinity but insufficient to break through the sand barrier, the wetland will experience a prolonged low-salinity gap.

3) Invasions that follow substrate disruption

The periphery of most California salt marshes supports mixed vegetation of native and introduced species. The transition from wetland to upland is often marked by an increase in species richness and an abundance of exotic plant species. Their origins, according to Munz (1974), include Europe (*Polypogon monspeliensis*), Eurasia (*Bassia hyssopifolia, Salsola iberica*), Chile (*Cortaderia atacamensis*), South Africa (*Gasoul* [*Mesembryanthemum*] species), and Australia (*Atriplex semibaccata*). [Note: Abrams (1951) states that *M. nodiflorum* probably arrived in California before European man.] They are not necessarily weeds in their home country. In England, for instance, *Polypogon monspeliensis* is valued as a rare plant and its habitats are actively managed to maintain populations (A. J. Gray, pers. comm.).

These exotic species mark areas where the coastal soils have been disturbed by agriculture or horticulture, dumping, sediment plumes, street drains, spoil disposal from dredging operations, trampling or vehicle use. They are also able to invade sites of natural disturbance, such as animal burrows (Cox and Zedler 1986), slope failures, and alluvial fans. The herbaceous species may also develop where wrack (tidal debris) is deposited and marsh vegetation is smothered, so long as the elevation is very high in the intertidal zone (e.g., extreme high water). These peripheral invaders obviously have some salt tolerance, but their inundation tolerance is limited. Many are restricted to coastal habitats, including bluffs and the salt marsh periphery, which suggests either a salt requirement or poor competitive ability in low salinity soils. In southern California, *Atriplex semibaccata* is a common dominant of the marsh-upland transition (Ferren 1985, Cox and Zedler 1986, Zedler and Nordby 1986).

The list of alien weeds would no doubt be lengthened by studies of the high marsh and transition to upland throughout California. A variety of annual grasses are usually found there, but complete species lists are lacking. This transitional area is the most poorly-known habitat of our coastal wetlands. Most of its area has already been destroyed, and what remains has been badly disturbed. Thus, an understanding of the native plant community and how it is affected by exotic species may never be possible.

DISCUSSION AND RECOMMENDATIONS

The problem of exotic species invasions extends to the entire coastline and to all coastal habitats. The environmental modifications that facilitate the expansion of weedy species are equally widespread. Hydrologic modifications are increasing in magnitude and area affected. Demands for water in the southern part of the state, if satisfied, will jeopardize inflows into San Francisco Bay. With new water supplies to the arid southwest will come new problems with wastewater disposal. If inland municipalities shift their wastewater discharges from ocean outfalls to coastal rivers, the streamflow regimes for coastal water bodies will be permanently altered. Flood flows are necessarily prolonged if flood control dams are to perform their function of reducing peak flows. Where urban runoff flows directly into salt marshes, alien species will follow the deposition of sediments and the influence of fresh water. Wherever salt marsh sods are disturbed, problems with exotics will likely develop. In general, the higher marsh and buffer zones are more susceptible to invasion than the lower marsh, because few species can tolerate frequent submergence by seawater, and few species can germinate and establish seedlings in saline to hypersaline soils. Combinations of disturbances, such as alterations to hydrology plus disrupted soil, improve the opportunities for seed germination and seedling establishment, because the competitive ability of the native plants is reduced where roots and rhizomes have been severed. At the same time that coastal landscapes are being modified, seed supplies of exotic species are accumulating. With expanding opportunities for invasion and an increasing source of exotic seeds, the potential for species invasions increases dramatically.

Control measures become essential whenever the invader is considered noxious or the affected resource is highly valued. In California, all of the coastal wetland habitats are valued, not only as native ecosystems but as habitat for rare and endangered plants and animals. Although the problems of exotic plants in wetlands have not been well documented, there is sufficient evidence and opinion that controls are needed.

The best time to control an invasive plant species is before it gains dominance. The native community will suffer less damage, and the disturbance caused during control activities will affect a smaller area, thus reducing chances for weed reestablishment. Experiences with wetland weed control are limited, and success stories are few. What works in an agricultural field, namely cultivation and herbicide application followed by planting of the desired crop, does not work in native wetland communities. Efforts to eradicate *Chrysanthemum* from filled areas at Tijuana Estuary by disking only retained the disturbed-soil conditions that facilitated its dominance (pers. obs.). Controlled burning to eliminate seeds and accumulated biomass is often ineffective (W. Tippets, pers. comm.); if done in the driest weather, it may be hazardous to adjacent developments. Seed mixes to enhance germination of native halophytes are unavailable from nurseries. Hence, hand gathering and sowing is required, and problems should not be left until large-scale operations are necessary. Early diagnosis and treatment are essential, and hand tools rather than machines are desirable. Where herbicides are necessary, spot application by hand is preferable to broadcasting.

Preventative measures can help reduce problems. Controlling street runoff to salt marshes is imperative, not only to stop the year-round freshwater inflows but also to keep toxic materials and sediments from being dumped in the marsh. Upstream sediment traps are certainly needed; hook-ups to sewer lines would offer better protection for the wetland. Where large drains enter estuarine channels and sewer connections are not possible, controls on the source of the problem may have to yield to treatment of the symptoms. In such situations, it is important that good tidal flushing and rapid dilution be maintained.

A simple remedy for horticultural escapes is the planting of native species in buffer zones adjacent to wetlands. What limits their use appears to be a lack of knowledge among planners and landscapers. An education program is needed. The objection that "native species aren't as esthetically pleasing" can be negated using photos of shrubs with open growth form and masses of flowers. The problem that planting methods are uncertain can be met by experimental seeding and transplanting trials. The complaint that nurseries lack adequate stock can be met once increased demand drives supplies. The desirable attributes of native plants for landscaping transitional habitats need to be publicized, and a demonstration project should help to change attitudes. For instance, *Juncus acutus* (spiny rush) is an effective deterrent to cats and dogs; a border of these sharp-leaved plants is nearly impenetrable. The drought and salinity tolerance of *Malosma (Rhus) laurina* makes it carefree following initial establishment. The preferences that native insects have for native plants over exotics such as *Carpobrotus edulis* needs to be made known.

Finally, to reduce problems with the transition-habitat exotics, it is important to maintain the integrity of native sod. The widespread use of disking, tilling, and grading topography should not be considered for weed control. Instead, native vegetation should be encouraged to dominate the site by small-scale soil preparation (patch rototilling or augering) and planting. Appropriate techniques are being developed in an experimental restoration program at Tijuana Estuary, including tests of the relative effectiveness of transplanting and seeding high marsh plants (separately and in combined treatments) into areas that were denuded by off-road vehicles (M. Weitzel, U.S. FWS Refuge, Imperial Beach, pers. comm.). The results of these small-scale patch trials will be compared to previous disking operations (without planting) where exotic species were quick to dominate and native species were slow to colonize. In addition to stimulating growth of exotics, the disking operations altered soil topography and impounded rainfall, so that unnatural "furrows" of wetter soil developed in the high marsh. Development of high marsh-transition vegetation on newly graded topography will prove difficult, and experimental work to create mixed-species sods for transplantation to the high intertidal zone has barely begun (B. Fink, PERL, pers. comm.).

There is much room for research work concerning the habitat requirements of wetland exotics, use of exotics by native animals (especially insects), effects of exotic species on the soils, native-exotic species interactions, and control measures. It is certain that more is known about these important species than has been summarized here. So much of the knowledge of our wetlands exists in the experience of managers and naturalists.

In conclusion, the control of exotic species invasions in coastal wetlands is a necessary but difficult task. Existing problems need to be controlled and future expansions restricted. Small-scale measures that minimize disruption to native sods are recommended, with continual surveillance to slow reestablishment. It is overly optimistic to aim for eradication of species that are so invasive and widespread; however, substantial reduction of existing problem areas and preventing spread to new sites are reasonable goals. Wherever there are uncertainties in how to approach the problem, an initial small-scale experimental approach is recommended.

ACKNOWLEDGEMENTS

Paul Kelley of the California Department of Fish and Game commissioned the original version of this paper; I am grateful for his interest in the topic. Discussions with several people made me aware of the seriousness of the problems with species that invade coastal wetlands, especially Rob Atkins and Arthur McComb in Perth, Western Australia; Tom Mumford, Barbara Aberle, and others concerned about *Spartina* invasions in the Pacific Northwest; and Mike Josselyn, who is concerned about exotic *Spartina* species in San Francisco Bay. Alan Gray and anonymous reviewers improved the text. I am especially grateful to Pam Beare, who figured out why *Typha* invaded the San Diego River salt marsh, and to Eric Paling, without whom the *Typha* invasions around the Swan River Estuary would still be a mystery.

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Harvest Refuges And Their Potential For Enhancing Reef Fisheries In Southern California

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Abstract[1]. The use of harvest refuges to manage and protect nearshore fisheries has been receiving increasing attention throughout the world including southern California [1, 2]. Harvest refuges are based on the premise that populations protected from harvesting act as a spawning source from which larvae disperse to and replenish harvested populations. Thus harvest refuges might provide better protection and/or enhancement of fisheries relative to more conventional management strategies. Nonetheless, harvest refuges should supplement, not replace, current management practices such as size and catch limits and limited entry fisheries. Assemblages of temperate reef fishes are among the most valuable resources subject to multiple use conflicts in urbanized coastal zones like southern California. Having a protracted planktonic larval stage, many reef fishes are composed of populations in which local recruitment is decoupled from local adult fecundity. Such "open populations" (contrasted with "closed populations" in which recruitment is largely dependent on local spawning stock) may be particularly well suited for management by harvest refuges because of the greater potential of these populations to augment production of stocks located outside of the refuge within the broad range of larval dispersal. In this paper we address several conceptual issues relevant to the design, management and performance assessment of harvest refuges as a means of enhancing reef fisheries in southern California. We point out the striking differences among species in their modes of larval replenishment, resource requirements, and habitat use and we discuss the implications of these differences to the design and management of the harvest refuge.

HARVEST REFUGES

Detailed information on the local oceanography and on the biology and ecology of the target species is essential to the successful design of harvest refuges. Design criteria for harvest refuges include their size, number, distribution and habitat type(s). Of fundamental importance in determining refuge size is the number of larvae needed to sustain a harvested population at some predetermined level of fishing. Obviously the number of larvae produced by a refuge will depend on its areal size and adult density. The number and distribution of harvest refuges necessary to replenish an exploited population depends on the geographic range over which a particular refuge can supply new recruits. This range is determined largely by factors intrinsic to a species (e.g.

[1] A more detailed version of this paper will appear in the *Canadian Journal of Aquatic Sciences and Fisheries* as part of a symposium on harvest refugia presented at the 121st Annual Meetings of the American Fisheries Society.

mode of reproduction, timing and location of spawning, and developmental and behavioral characteristics of larvae) as well as extrinsic environmental factors that influence the duration, distance and direction of dispersal of pelagic stages of the target species. Perhaps the most important extrinsic factor affecting patterns of larval replenishment is ocean currents. There is marked seasonal variation in currents in the Southern California Bight that result from a cross-shore shift in the Southern California Gyre. In the fall the gyre occurs offshore and the northward flow of the Davidson Current inshore strengthens as it moves to the surface. In contrast, the gyre moves inshore in the spring and the bight is dominated by the strong southerly flow of the California Current. Such seasonal shifts likely influence the direction and distance of larval dispersal depending on the season and location in which fish spawn. For instance, kelp bass (*Paralabrax clathratus*) and shallow-dwelling rockfishes (genus *Sebastes*) spawn during the summer and winter periods, respectively. Therefore, larvae produced by adults of these taxa within the same refuge might be dispersed very different distances and directions. Large scale interannual variation in currents is also common in southern California. During El Niño events, for example, the gyre weakens and relatively warm water from the south flows northward throughout the bight. In contrast, during anomalous cold water years (i.e. La Niña) the Southern California Gyre moves farther inshore and cold northerly water of the California Current flow south throughout the bight. These anomalous conditions can persist for several years and can dramatically alter ranges of larval replenishment. For instance, recruitment of blue rockfish (*Sebastes mystinus*) into the bight occurs during cold water years such as those corresponding with La Niña episodes [3]. Apparently, larvae can be transported down into the bight from major spawning sources north of Point Conception. In contrast, recruitment of sheephead (*Semicossyphus pulcher*) to populations in the northern portion of the bight corresponds to El Niño events [4]. Larvae appear to be transported to the north from major spawning sources south of the bight by the northward flowing currents during El Niño events. Therefore, refuges for these two species should be located at opposite ends of the bight at their respective major spawning sources.

Of critical importance to the successful design of harvest refuges is the delineation of the effective spawning stock(s) of the populations considered for enhancement. Patterns of larval replenishment among local reef fish populations are generally typified by one of four models that characterize the different replenishment patterns exhibited by most reef fishes in the Southern California Bight.

(1) Closed population model. This model typifies species whose local populations replenish themselves but contribute little to the replenishment of other populations. Such populations are characterized by species with very limited offspring dispersal or whose offspring, although dispersed, return to breed with members of the same population from which they were spawned. Examples include the reef-associated surfperches (family *Embiotocidae*) which give birth to well developed young that have no planktonic stage [5]. Since offspring are not dispersed from parental populations harvest refuges are simply not appropriate because adults protected within a refuge would contribute little to the replenishment of harvest populations outside the refuge.

(2) Single source model. In this model populations are replenished primarily by a single spawning stock (or "source" sensu [6]) that supplies larval recruits to populations throughout the Southern California Bight; non-source populations (or "sinks" sensu [6]) generally contribute little to their own replenishment or that of other populations. Possible candidates for this type of larval replenishment might include the sheephead (*Semicossyphus pulcher*) [4] and blue rockfish (*Sebastes mystinus*) [3] whose populations appear to be replenished by major spawning sources to the south and north of the bight, respectively. For species with populations that rely on a single source for replenishment, harvest refuges must be located at the major source of larval supply.

(3) Multiple source model. This is the classic metapopulation model in which several isolated breeding populations contribute to a common larval pool from which each isolated population is eventually replenished. Species that typify this model have relatively long-lived larvae that are capable of dispersing to any subpopulation in the Southern California Bight. Possible examples include the bocaccio (*Sebastes paucispinis*) [7], halfmoon (*Medialuna californiensis*) [8], and opaleye (*Girella nigricans*) [8,9], whose larvae and pelagic juveniles are thought to spend long periods in the plankton. An important concern in the design of harvest refuges for such species is to distribute the refuges throughout the bight so as to spread the risk of losing all refuges at any one time.

(4) Limited distance model. Species whose larvae are in the plankton for relatively short durations such as kelp bass (*Paralabrax clathratus*) and shallow dwelling rockfish (e.g. *Sebastes atrovirens, Sebastes carnatus*) may have limited larval transport. Consequently, neighboring populations may exchange more larvae than distant populations. For such species it is necessary that refuges be placed within the range of replenishment of nearby refuges to guarantee replenishment of one another as well as the harvested populations between them.

The four models are not mutually exclusive. For instance, models 2 and 3 can be subject to constraints of limited larval dispersal (model 4) in some cases, and a major spawning source might be comprised of several local populations (model 3) that together contribute to episodic replenishment of other populations (model 2). As suggested by these four population models, differences in modes of larval replenishment will require that the number and distribution of refuges differ among species. Determining which population model best describes the mode of replenishment of a given target species is not a simple task. However, this information is crucial to the successful design of harvest refuges. Modern techniques in molecular biology may prove to be invaluable tools for distinguishing spawning stocks. The ability to use genetic polymorphisms in proteins and nucleic acids to identify cohorts originating from populations subjected to different hydrographic regimes, coupled with adequate knowledge of oceanography and of the biology and ecology (e.g. larval duration and dispersal potential, cohort strength, migration) of the target species, may be extremely useful in determining the most likely sources of replenishment under different oceanographic conditions.

Of fundamental importance in determining the area and habitat type within a refuge is that resources required by all life stages be available within, or in close proximity to, a refuge. Many reef fishes exhibit marked changes in resource requirements over their lifetime and these changes often result in shifts among habitats during ontogeny. For example, kelp forests are inhabited by a diverse assemblage of reef fishes. Both juvenile and adult stages of some species such as the kelp bass (*Paralabrax clathratus*) associate with kelp [10]. However, many of the commercially important rockfish, such as the bocaccio (*Sebastes paucispinis*) occur in kelp forests only as juveniles whereas adults inhabit deeper reefs offshore [11]. Further, adult opaleye (*Girella nigricans*) and halfmoon (*Medialuna californiensis*) inhabit kelp forests as adults but their young occur primarily in shallow intertidal and pelagic habitats, respectively [8,9]. Consequently, refuges that encompass a variety of habitat types may be required to include the breadth of resources required by all life stages of a targeted species.

Recognition of a targeted species' resource requirements also has important implications for proper refuge management. An important consideration in the managing of harvest refuges is the type and degree of interactions between the target species and other species in the refuge. Within refuges, reef fishes, invertebrates, and algae all interact and influence one another in both positive and negative ways. Protecting species beneficial to the target species and/or allowing the harvest of species that are detrimental to the target species may enhance the quality of the refuge for the target species. In addition, it may be necessary to protect resources outside the refuge that enhance recruitment of the targeted species into the fishery. For example, giant kelp (*Macrocystis pyrifera*) is known to increase the local density of larval recruits of a diverse group of reef fishes [11, 12].

One of the most important responsibilities of management will be in evaluating the effectiveness of a refuge in augmenting populations of particular targeted species. Refuges that do not contribute to fishery enhancement are not only a waste of time, effort and money but also restrict the range of the fishery unnecessarily. Further, establishing refuges does not guarantee the protection of refuge or harvested populations. Overfishing harvested populations could endanger refuge populations if refuges rely on replenishment from unprotected areas. Therefore it is imperative that a refuge be evaluated on its ability to: (1) increase and maintain the abundance of the target species in the refuge relative to harvested areas (i.e. to be self replenishing), and (2) contribute to the replenishment of harvested populations at a level sufficient to sustain a predetermined fishing effort. The evaluation of these criteria will most likely not be an easy task. Rigorous long-term monitoring of populations of targeted species in harvested and non-harvested areas should be an essential and integral part of any harvest refuge plan. Such monitoring should be done both before and after the refuge(s) has been established [e.g. 13]. In addition to monitoring recruitment to harvested populations it will also be necessary to measure the relative contribution of the refuge to harvested stocks. The problems and methodologies in determining this contribution are akin to those encountered in assessing the contribution of hatchery programs to fisheries [14]. Molecular techniques that identify the origin of recruits may be useful in assessing this contribution. These modern methods may only prove useful, however, when coupled with a detailed understanding of the biological and physical processes affecting successful recruitment of the targeted species. For example, studies that integrate oceanographic, ichthyoplankton and juvenile recruitment data would provide valuable information for interpreting studies of population genetics.

In conclusion, most reef fishes differ with respect to several factors that determine the successful design of a harvest refuge. Species often differ with respect to modes of population replenishment and they differ with respect to their resource requirements and the way that changes in resource requirements during ontogeny lead to differences in habitat use. The primary implications of these conclusions are: (1) the success of a refuge will depend on our understanding of the mechanisms of population replenishment and patterns of resource use of a targeted species, and (2) although logistically and economically appealing, the success of a single refuge for many species is unlikely given the many differences among reef fishes.

ACKNOWLEDGEMENTS

We thank A. Ebeling, S. Holbrook and M. Franklin for their ideas and comments and the National Science Foundation (OCE88-33931) and the University of Southern California Sea Grant Program (NA 86AA-D-SG119) for support.

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Habitat Value Of A Southern California Artificial Reef

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Abstract. Artificial reefs are one of the few potentially useful means of compensating for losses of coastal marine biological resources to development. Their use, however, has been hampered by uncertainty concerning the extent to which artificial reefs actually produce fish rather than simply attracting fish from elsewhere, and the lack of a way to compare the ecological value of a reef to that of the habitat for which it would serve as compensation. The present study, sponsored by the Port of Long Beach, the National Marine Fisheries Service, and the Port of Los Angeles, was designed to address those two issues.

The problem of measuring fish production on an artificial reef is complicated by the diversity of life styles and forms of production of the organisms under consideration. To address this difficulty, the study used a variety of techniques and analytical approaches for estimating abundance and production. Fish standing stocks and production were estimated on and near Torrey Pines Artificial Reef (TPAR), a 0.18-ha. quarry-rock reef in San Diego County, during a nine-month study in 1989. Fish standing stocks were estimated in two basic ways: by analyzing mark-resighting data from tagging studies performed at the beginning and end of the growth season (the Schnabel technique), and from visual census data collected by divers. Tagging started in April, 1989; 457 fish of eight species were tagged. In October and November, 1989, a second tagging study provided an estimate of density at the end of the study for the tagged species, and also provided a measure of short-term losses due to the tagging itself. In addition, fish tagged in April and May were recaptured at this time for measurements of seasonal growth rates.

Thirteen visual censuses of older fish abundances, length-frequency distribution, and tag ratios, and 14 surveys of young-of-year and small cryptic fish were conducted between May and November, 1989. Divers conducted quantitative replicate transect censuses in three reef strata (crest, slope, sand-rock ecotone) and in three sand strata defined by distance (up to 30 m) from the edge of the reef. The sand-bottom fish assemblage was assessed by monthly trawl sampling approximately 0.5 km from the reef.

Somatic production was calculated in three ways; (1) for six of the tagged species (garibaldi, black surfperch, rock wrasse, sheephead, kelp bass, barred sand bass), by extrapolation from the measured growth of tagged and recaptured individual fish and average abundances as defined by mark-recapture data; (2) for blacksmith, from the growth in mean length of a cohort over time; and (3) for the remaining species, from the change in total population biomass over the course of the study. Gonadal production of six target species (five of the six tagged species, plus blacksmith in place of black surfperch, which is a live-bearer) was estimated from analyses of gonadal indices of fish captured at Pendleton Artificial Reef (PAR), approximately 60 km northwest of TPAR.

The average abundance of larger fish over TPAR proper, estimated from visual census data, was 1071 individuals, or 5951 per hectare. Densities were markedly greater over the reef crest than in any other habitat or stratum (Figure 1). The density of young-of-year and cryptic species combined was about an order of magnitude greater. Densities of fish over the sand bottom, by contrast, averaged about 544 per hectare. Among larger fish, blacksmith was by far the most abundant species, comprising some 30 to 50% of the individuals. Of the other target older fish, garibaldi and kelp bass were consistently abundant. Senorita was the most abundant non-target older fish, with up to 450 individuals present on the reef. Estimated abundances of small fish — young-of-year and small cryptic fish such as gobies — on the reef itself averaged 9700, but varied over two orders of magnitude, between 170 and 25,589. Young-of-year blacksmith comprised 90% of the small fish during most of the study, but young-of-year senorita and bluebanded and blackeye gobies were also very abundant at times.

Calculated somatic production of reef fish during the study period ranged from 1.1 kg for rock wrasse to 16.3 kg for kelp bass (Table 1). The estimate for blacksmith (12.8 kg) does not include the growth of parts of the population not included in the measured cohort, and is thus a substantial underestimate. Production by non-target species, calculated from the change in population biomass, was substantially smaller than production by the target species, ranging from 3 kg for painted greenling down to only 0.3 kg for bluebanded goby. This is because the list of target species included all of the large, abundant species that would contribute the most to production. The estimate of total production did not, however, include the night-active community, including scorpionfish, which was known to be extremely abundant at TPAR. Gonadal production by the six target species for which it could be estimated totaled 46.6 kg, over half of it due to the two bass species. Total production, somatic and gonadal, was approximately 116 kg, equivalent to 646 kg per hectare (Table 1).

Production of fish over the open sand bottom was estimated from abundance data collected in 1980-1981 (DeMartini and Allen, 1983) because the absence of the normally dominant croakers from the 1989 samples suggested that 1989 was an atypical year. Those data and literature-derived P:B and gonadal output data suggested that the production of the sand-bottom fish assemblage was approximately 73 kg per hectare. (For 1989, the estimate was 28.8 kg per hectare). On a per-area basis, therefore, production by reef fish was approximately nine times that of sand-bottom fish.

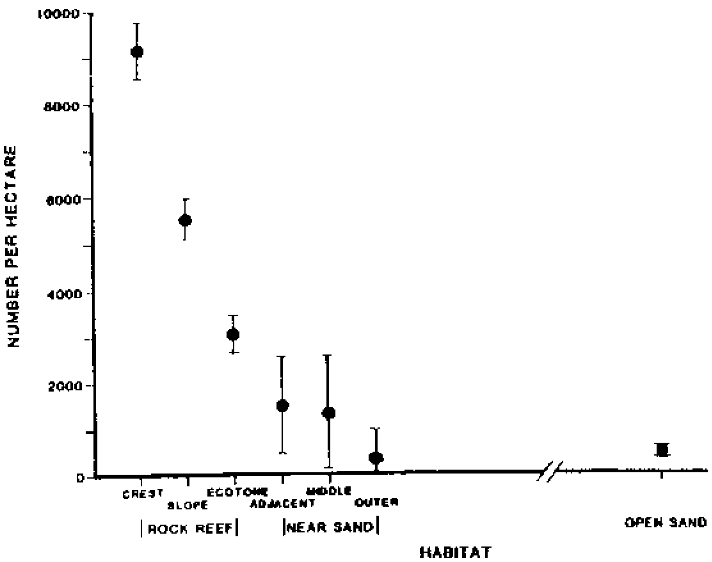

Figure 1. Density of Older Fish

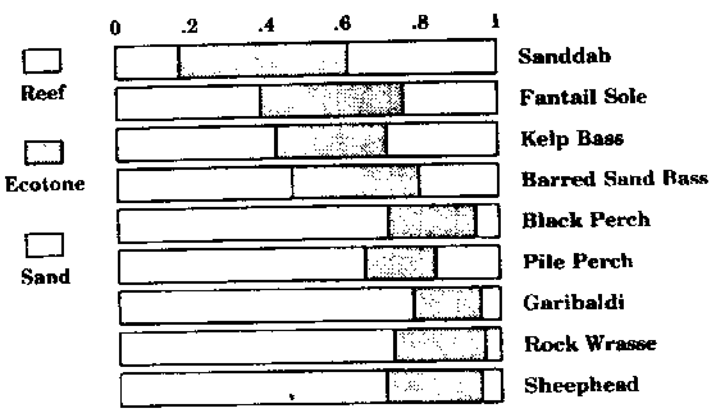

Figure 2. Proportion of Diet from Each Habitat

The recapture rates of tagged fish at the end of the study ranged from 13 to 70 percent, depending upon the species (Table 2). The high recapture rates for the species other than basses indicate a high degree of fidelity to the reef. The growth studies showed that the fish did grow during the time they were on the reef. A comparative, quantitative taxonomic and biomass analysis of fish stomachs and benthic samples, based upon the Benthic Resource Assessment Technique (Lunz and Kendall, 1982) as modified by MEC (1988), showed that sand-associated and reef-associated species fed predominately in their respective habitats. The former (sanddab, lizardfish, and, to a lesser extent, the two basses), derived at least 60% of their diets from the sand and sand-rock ecotone, whereas reef-associated fish (the wrasses, garibaldi, and surfperches) derived at least 70% of their diets from the rock (Figure 2). Overall, suitable food for most of the target species was approximately two orders of magnitude more abundant in the reef habitat than in the sand habitats. Thus, the reef represents a valuable living space and food resource for reef species.

An ecologically-based, quantitative valuation method called BEST, developed by MEC, was used to compare the ecological values of three habitats. The method uses a suite of target species — in this case the target species for the reef study and ecological equivalents in soft-bottom habitats — to express ecological value as defined by the use of habitat by the target species for living space, nursery area, and a food resource. A measure of total fish productivity was also incorporated. The value of the reef was first compared to that of the open sand bottom which it would replace, then the value of the reef minus the value of the sand bottom was compared to the value of deep-water harbor habitats for which the reef would be used as mitigation. The comparisons showed the ecological value of the reef to be nearly six times that of the nearby sand bottom. The value of the reef adjusted to account for the value of the sand bottom that would be lost by reef emplacement was over 1.5 times the value of the areas of Los Angeles/Long Beach Harbor for which dredging and filling are planned. That estimate is thought to be conservative — i.e., it is likely that the relative ecological value of the reef is actually higher — because the production value for the reef was based upon a subset of the total number of fish present and on a seven-month "growing season," whereas those for the harbor and the open-sand bottom were based upon all of the fish and upon the entire year.

This study is important in showing that artificial reefs have inherent ecological value. The study was also able to compare that value to the ecological value of other habitats and to show that value to be relatively high. Thus, the study has shown that artificial reefs have considerable potential as compensation for developments in coastal marine waters.

Table 1

Summary of Production by Reef Fish, May-November 1989

(Kg. on the reef)

Species	Somatic	Gonadal	Total
Painted greenling	3.1	—	3.1
Kelp bass	16.3	12.6	29.0
Barred sand bass	3.1	16.3	19.5
Black surfperch	3.6	—	3.6
Blacksmith	25.3	6.6	31.9
Garibaldi	5.0	5.4	10.4
Rock wrasse	1.1	3.6	4.7
Senorita	3.4	—	3.4
Sheephead	7.4	2.1	9.5
Blackeye goby	0.7	—	0.7
Bluebanded goby	0.3	—	0.3
Totals	**69.7**	**46.6**	**116.5**

Kg. per hectare: 648.8

Table 2

Numbers of Fish Tagged and Recaptured at TPAR

Species	Tagged (Apr-May)	Recaptured (Oct-Nov)	Percent
Kelp bass	131	16	12
Black surfperch	99	39	39
Garibaldi	91	27	30
Rock wrasse	45	24	53
Sheephead	51	36	71
Others*	40	0	0
Totals	**457**	**142**	**31**

*Pile perch, California halibut, Barred sand bass

ACKNOWLEDGEMENTS

This study was conducted by MEC Analytical Systems, Inc. and supported by the ports of Long Beach and Los Angeles and by the National Marine Fisheries Service through Grant No. NA89AB-H- SK019 under the Saltonstall-Kennedy program

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Implications For The Design Of Environmental Assessment Studies

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Abstract. Many types of environmental impact assessment studies aim to detect effects of localized impacts. Most of the assessment (or compliance monitoring) designs used in such studies fail to distinguish effects of anthropogenic versus natural origins, and thus might lead to incorrect interpretations. The Before-After- Control-Impact-Paired (BACIP) design surmounts this and other problems, yet has rarely been used in assessment studies. For the past three years, we have used BACIP to study possible effects of nearshore discharge of produced water, an aqueous waste generated during oil production. Results from power analyses suggest that environmental impacts are more likely to be detected for physical and chemical parameters than for biological measures; within biological parameters, effects on individual-based properties (e.g., growth, fecundity) are more likely to be detected than changes in population densities. However, regulatory agencies and resource managers ultimately are concerned with impacts on populations and communities. Our results emphasize the need to: (i) collect adequate (time-series) data before a localized perturbation begins, (ii) understand mechanisms that lead to population change and (iii) develop comprehensive models of processes leading to environmental impacts.

INTRODUCTION

There continues to be considerable debate regarding localized effects of anthropogenic disturbances on marine biotic resources. Controversy arises, in part, from equivocal data obtained from poorly designed environmental assessment studies. Despite great efforts to obtain reliable information, most assessment designs currently in use fail to provide rigorous and convincing tests of possible effects [1]. Most of these studies fall short because the assessment designs employed do not separate changes in ecological systems caused by the putative impact from changes resulting from natural spatial and/or temporal variation [1,2,3,4]. Consequently, it is often difficult to draw scientifically defensible conclusions about the existence or magnitude of a localized environmental impact.

Even when seriously flawed, many assessment designs may be sufficient to "demonstrate" large, severe impacts. However, such dramatic (qualitative) changes in marine ecosystems are unlikely to be common due to regulatory steps such as strict permitting conditions and relatively stringent monitoring of effluents. Of more concern are long-term chronic effects, which can involve quantitative changes (e.g., reductions in population abundance) and which can accrue slowly. By their nature, chronic impacts are more insidious, less easily isolated from natural variability, and therefore demand more rigorous assessment designs to detect. Few assessment studies have used such rigorous designs, and as a result, regulatory agencies typically lack the sufficient scientific information to make environmentally sound decisions concerning management of marine resources.

In this paper, we begin by discussing the goal of environmental assessment studies, and then illustrate how the three most commonly used assessment designs fail to satisfy these goals. We review an alternate design, which controls for many types of natural spatial and temporal variation and therefore provides a more defensible approach to environmental impact assessment. This alternate design reduces the possibility of wrongly concluding that an impact has occurred. However, a concern generic to all assessment designs — that actual impacts might go undetected — still remains whenever statistical power is low. We consider the issue of power, and suggest that the ability to detect actual effects may vary systematically with the type of parameter measured. These results have fundamental implications for interpreting results of all assessment studies.

We illustrate many of our points with data from our ongoing study of possible environmental effects of nearshore discharge of "produced water" on benthic marine organisms. Produced water is an aqueous waste generated during oil production, and is contaminated with various petroleum hydrocarbons, heavy metals and other inorganic chemicals, as well as additives (including biocides) introduced to increase the separation of produced water from crude oil [5,6]. Since January 1988, we have been conducting a detailed study to assess whether environmental impacts result from the discharge of produced water. The system we focus on is a soft-bottom community occurring near Gaviota, California at a bottom depth of approximately 25 m. Although the produced water study is specific in its primary intent, the message of this paper is relevant to the study of localized environmental impacts in general.

THE GOALS OF ENVIRONMENTAL IMPACT ASSESSMENT

In general, the question to be answered by an assessment study of a localized perturbation (e.g., wastewater discharge) is: "How does the ecosystem at the site of perturbation differ from the ecosystem that would have existed had the perturbation never occurred?" Obviously, the answer cannot be obtained by direct observation, and the goal of an assessment design should be to estimate the state of the system that would have existed in the absence of the perturbation [7]. Further, this estimate should be statistically compared to the observed condition (in the presence of the perturbation) and a probability should be assigned that the estimated effect might have arisen by chance (i.e., due to natural variability in the absence of an impact). If a statistically significant result is not obtained, it is absolutely critical to estimate the "power" of the test, which is the probability that the analysis could have detected an impact had it occurred.

Table 1

Two types of errors committed in environmental assessment studies

Type of Error	Conclusion	Reality
False Implication	"Impact"	None
False Exoneration	"None"	Impact

There are two types of errors that can be made in interpreting results from an assessment study (Table 1). We call the first type of error, "False Implication." It arises when we conclude that a perturbation has resulted in an environmental impact when in reality the effects we see arose for another reason (e.g., due to natural variability in the system). The second type of error is "False Exoneration," in which we conclude there has been no impact, but in fact there has been one. The first error might result in unnecessary regulation of environmentally safe projects, while the second might fail to alert regulators to environmental impacts that require prevention or mitigation. Each type of error can have serious implications and should be minimized within constraints imposed by the study. We now consider the relative merits of several impact assessment designs.

THREE COMMON ASSESSMENT DESIGNS AND THEIR LIMITATIONS

A widely used assessment design, often employed in compliance monitoring in the state of California, is one in which an impact site (or a gradient of impact sites) is sampled and compared to a more distant control site(s) after a perturbation has begun. We refer to this as the "Control-Impact" design. Differences in parameters of interest (e.g., population densities) between the sites are taken to represent effects due to the perturbation. However, ecological systems exhibit considerable spatial variation and it is not possible to reliably interpret any difference between sites as being due to the perturbation: differences might exist for a number of possible reasons. For example, we have estimated densities of a large epifaunal gastropod (***Kelletia kelletii***) at our Gaviota study sites. Figure 1 shows densities at two impact sites (50 m and 250 m downcurrent from the diffusers) and at a control site (1500 m upcurrent from the diffusers). Clearly, gastropods were less abundant at the impact sites, and it might be concluded that produced water discharge negatively affected gastropod density. However, at the time these data were collected, produced water had never been discharged. In fact, the differences in densities between control and impact sites (Figure 1) were simply the result of other processes that led to spatial variation among the sites. Had these data been collected after discharge, the "Control-Impact" design could have led to the false implication of an impact (Table 1).

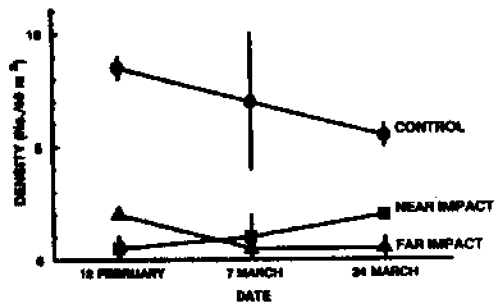

Figure 1 . Density of the gastropod, *Kelletia kelletii*, at three sites over time. The near (square) and far (triangle) impact sites are located 60 m and 250 m downcurrent of a produced water outfall respectively, and the control site (circle) is 1500 m upcurrent. No produced water had ever been discharged at these sites when the data were collected. Shown for each date are the mean and range of gastropod density (N = 2 band transects per site).

A second approach compares the condition of an impact site before the perturbation occurred with the condition of the site after the perturbation. This we call the "Before-After" design. Although this design circumvents problems associated with natural *spatial* variation (as discussed above), it instead ignores natural *temporal* variation, which is also ubiquitous in nature. To illustrate this problem we use data collected by the Marine Review Committee in a study of the San Onofre Nuclear Generating Station, SONGS [8,9]. Densities of pink surfperch were estimated over time before and after new units at SONGS began generating power [8]. The density of pink surfperch declined markedly, with the reduction coinciding with the commencement of power generation by the new units (Figure 2). It is tempting to conclude from these data that the operation of the new units (accompanied by discharge of cooling water) negatively affected the surfperch. However, these data were taken from a control site 18 km from SONGS. In reality, this temporal change in density occurred at all sites (control and impact alike), probably in response to El Niño [10]. Thus, in this case the "Before-After" design would have led to false implication because it failed to separate impacts from temporal variability introduced from natural sources.

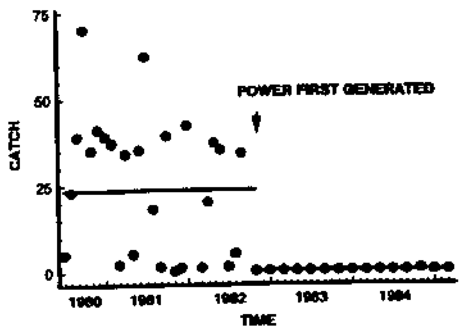

Figure 2. Density (catch per otter trawl) of pink surfperch, *Zalembius rosaceus*, over time at a location 18 km from the San Onofre Nuclear Generating Station (SONGS). The arrow indicates the first date on which power was generated by two new units of SONGS. Mean densities from the before and after periods are indicated by the solid lines.

One potential solution to the limitations of the "Control-Impact" and "Before-After" designs is to combine them into a single design in which control and impact sites are sampled both before and after a perturbation occurs. In this case, the test for an impact is conducted by asking whether the condition of the impact site relative to the control has changed from the before period to the after period. Green [11] proposed such a design, which he called the "Optimal Impact Assessment" design, but unfortunately recommended an inappropriate statistical test. He suggested using an error term based upon the observed error among all samples collected within a site during a particular period (e.g., replicate samples collected on a single date). For this test to work as designed, it requires the stringent assumption that differences in densities (or other parameters) between the control and impact sites remain exactly the same at all times. However, we know that sites exhibit unique temporal fluctuations under natural conditions. This natural variability, combined with sampling error, comprises the variation from which impacts must be distinguished. Green's design considers only the influence of sampling error. This shortcoming of the "Optimal Impact Assessment" design has been cogently pointed out by Stewart-Oaten et al. [12; see also 13], who noted that the design fails to separate local temporal variability of systems (which arise naturally), from long-term effects indicative of an environmental impact. For example, with sufficiently intensive sampling on two different dates (one in the Before and one in the After periods), use of within-site variation among "replicates" will always yield a significant relative change at the control and impact sites even in the absence of any anthropogenic disturbance.

To illustrate, we conducted such a test for data collected from our study of produced water impacts. The density of the seapen, *Acanthoptilum* sp., was sampled at both the control and impact sites during 1988 and 1990 (Figure 3). These two periods bracketed a projected date on which discharge of produced water was to begin and thus were expected to represent before and after conditions. An "Optimal Impact Assessment" test yielded a significant Site x Period interaction ($F_{1,78}$= 7.26, P< 0.01), suggesting that an impact had occurred at the outfall site. However, commencement of discharge was delayed and did not actually occur during this sampling interval. Therefore, in this instance the "Optimal Impact Assessment" design could have led to false implication.

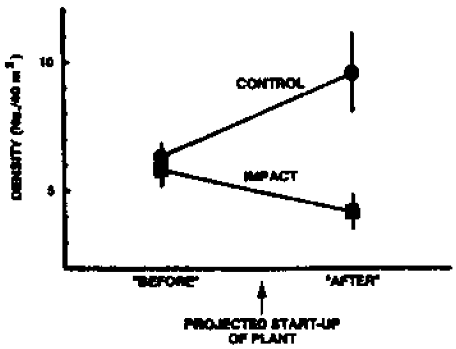

Figure 3. Density of the seapen, *Acanthoptilum* sp., at two sites. The control site is located 1500 m upcurrent, and the impact site 50 m downcurrent, of a produced water outfall. "Before" data are from 1988, and "After" data are from 1990; because of delays, discharge of produced water did not begin as expected between the two surveys. Shown are means (+ SE) using all observations within a period as replicates.

The central problem with the "Optimal Impact Assessment" design is that replicate samples (collected within a date or over a short time span) do not necessarily provide new and independent estimates of the general state of the impact or control sites. Instead, we require estimates obtained on many different dates sufficiently separated in time that data satisfy assumptions of independence. In other words, replication needs to be achieved through time (multiple sampling dates during the before and the after periods) and each replicate observation must be an independent estimate of the average environmental condition [12]. There is a fundamental lack of appreciation for this crucial aspect, yet it distinguishes a proper design from one with superficial similarity.

A PREFERABLE APPROACH
THE BEFORE-AFTER-CONTROL-IMPACT-PAIRED (BACIP) DESIGN

The fourth design we will discuss is the "Before-After-Control- Impact-Paired" (BACIP) design [12,14]. BACIP is somewhat similar in design to the "Optimal Impact Assessment" design, but it explicitly requires that sampling be conducted during several times in the before and after periods at both control and impact sites. For a given parameter (e.g., density), the variate of interest is the *difference* in a parameter value between the control and impact sites on a given date (e.g., population density at the control site minus density at the impact site). The measure of error in the statistical test is the variability of this difference, as assessed through repeated sampling in time. A number of assumptions must be satisfied to apply BACIP, and these assumptions (such as independence) have been rigorously elaborated by Stewart-Oaten and co-workers [12,14,15].

BACIP, relative to the other three designs, is most likely to isolate local impacts (e.g., from discharge of produced water) from natural sources of spatial and temporal variation. BACIP controls for the effect of spatial variation by measuring the average difference between the sites during the before period, and uses this difference as an estimate of the expected difference during the after period, assuming no impact. By focusing on differences, BACIP also removes the effect of temporal variation that affects both sites simultaneously (e.g., El Niño, winter storms). In essence, these temporal effects cancel upon subtracting the control and impact values. Finally, BACIP explicitly recognizes that sites fluctuate uniquely through time (i.e., exhibit Site x Time interactions) and therefore uses the variation through time in the difference between the control and impact sites as the estimate of error in the statistical test of an impact. This is the fundamental advantage gained by using a true BACIP design.

Despite its importance in environmental assessment, BACIP is relatively unappreciated as evidenced by its absence in recent discussions of assessment designs sponsored by National Oceanic and Atmospheric Administration and the National Science Foundation [1,13], the Environmental Protection Agency [16], the American Petroleum Institute [5], and the Minerals Management Service, the California State Water Resources Control Board, and the National Academy of Sciences [4]. Even though the development of this assessment design is rather simple and can be traced back to at least 1966 [17], BACIP has rarely been used or even discussed (but see 3,7,18,19,20). Unfortunately, fundamentally flawed designs such as the "Optimal Impact Assessment" design still motivate large, very expensive assessment programs [e.g., 21] and can lead to erroneous interpretation of environmental impacts.

Both false implication and false exoneration (Table 1) can be costly, and a well designed assessment study should explicitly address the commission of both types of errors. The probability of false implication is greatly reduced using BACIP (relative to the other designs discussed) because the impact is less likely to be confused with natural sources of variability. False exoneration remains a concerns with BACIP (as it does for any design) because there will arise situations in which there is insufficient evidence to statistically reject the null hypothesis of "no impact." In these situations it is tempting to conclude that there was in fact, no impact. In the absence of additional information, this potentially is a dangerous conclusion because there often can be substantial impacts that go undetected. Failure to detect such impacts arises when considerable variability in the system introduces a large error term in the statistical test. The probability of false exoneration is equivalent to the statistical Type II error rate (β). The power of a test is $1 - \beta$, which gives the probability of correctly concluding there has been an impact when an impact of a given size has actually occurred. This explicit specification of the power adds greatly to the interpretation of the test. We now turn to evaluation of power in our BACIP study of produced water effects.

STATISTICAL POWER

The power of a statistical test of an environmental impact (using a BACIP design) is influenced by four statistical attributes: (1) the Type I error rate (we assume here $\alpha = 0.05$), (2) the number of sampling dates (i.e., true replicates — the number of independent estimates of the difference between the control and impact sites), (3) the variability of these estimates (which we term S_Δ), and (4) the size of the impact that we wish to be able to detect. In general, power is high (closer to 1) when the number of survey dates (replicates) is large (Figure 4), the variability of differences (within the before and after periods) is low (Figure 4), and the anticipated impact is large.

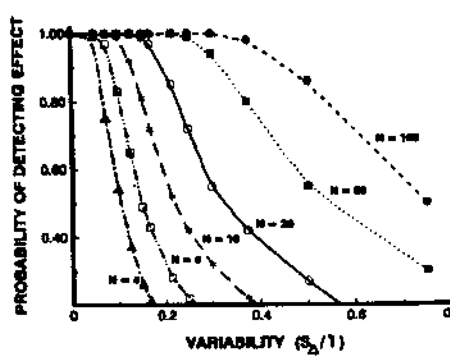

Figure 4. Effect of sample size and variability on statistical power to detect a 30% reduction (relative to control) in population density at the impact site. Given are power curves (after [22]) for 6 sample sizes (total number of sampling dates allocated equally to the before and after periods). Variability is expressed in a standardized form as the standard deviation of the differences between the control and impact sites (S_Δ), divided by the mean density at the impact site (I).

Because power analyses have rarely been applied to BACIP studies [but see 9], and because of continued confusion about the source of variability that is important in tests of impacts (see above), we illustrate the effect of high and low variability on power in Figure 5. In this case, we assumed that an anthropogenic perturbation caused a reduction in density at the impact site of 30%. In the left panel of Figure 5, we assumed that the difference between the control and impact sites varied considerably among sampling dates, while in the right panel we assumed that this variability was considerably less. As expected, it is more difficult to detect the 30% reduction in the case where there is high variability in the difference between control and impact sites (i.e., the power of the test is low). Notice that in both panels, the amount of temporal variability *within* a site is similar; the important distinction between these examples is the amount of variation expressed in the differences between control and impact sites. Further, nothing is assumed about spatial variation that exists within a date at a given site.

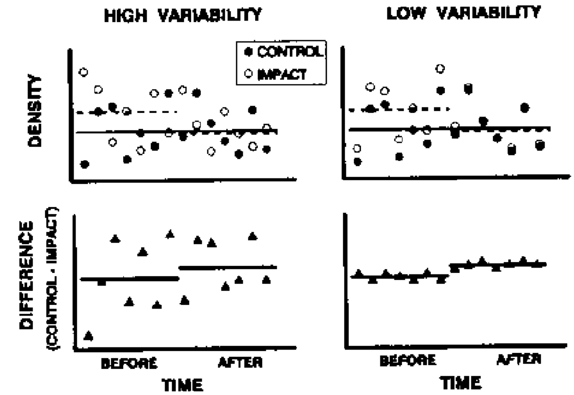

Figure 5. Effect of high and low variability in differences between the control and impact sites on the ability to detect an impact. The two top panels give hypothetical densities at control (solid circle) and impact (open circle) sites during the before and after periods; the bottom panels show the differences. Lines show means in each time period for control sites (solid), impact sites (dashed), and the mean difference (lower graph). Scales for left and right graphs are the same. Variance in densities at each site within a period is identical under the high and low variability scenarios. The degree of temporal consistency in the two sites differs between left and right graphs; under low variability, the control and impact sites are affected more similarly through time than under high variability. Although in each scenario the impacts are of the same size, high variability masks the impact; the impact can be much more easily detected under low variability.

The preceding discussion implies that greater power will arise if temporal changes in the value of a parameter track one another at control and impact sites. If qualitatively different classes of parameters (i.e., physical, chemical, biological) have consistently different patterns of this variability, the power of a test will depend on the particular type of parameter being examined. We have conducted power analyses for a number of parameters estimated at our near impact and control sites near Gaviota. These include population-based parameters (e.g., densities of macroinvertebrates and of infauna, emergence and re-entry rates of demersal zooplankton (estimated using methods of Alldredge and King [23] and Stretch [24]), individual-based parameters (e.g., mean body size, gonadal-somatic index), and physical and chemical parameters (e.g., sedimentation rate, percent organic matter in sediments, grain size of sediments). Here we summarize overall patterns, then illustrate specific conclusions using the white sea urchin, *Lytechinus anamesus.*

Our results indicate that, in general, power to detect impacts on population-level phenomena is relatively weak (compare Table 2 and Figure 4). The average (across species) variability in the difference in population density between sites was particularly large (Table 2), which greatly reduces power. For example, to detect an impact on the density of *Lytechinus* with 80% likelihood (assuming 25 sampling dates in each period), *Lytechinus* densities would have to decline (relative to the control site) by approximately 75%. Although *Lytechinus* provides one of the more extreme examples, power to detect impacts on population densities for most species we have examined also is low. There are, however, some species for which we have relatively high power (80%) to detect effects on density that are comparatively small — on the order of 25% (assuming 25 sampling dates each in the Before and After periods). The average variability for other population-level parameters (i.e., re-entry and emergence rates) was similarly large (Table 2).

As indicated by the lower variability for individual-based measures (Table 2), we have much greater power to detect impacts on such parameters as body size or gonadal-somatic index (GSI). To illustrate using *Lytechinus*, variability in density was approximately 1.5, while variability in mean test diameter was only 0.04. In general, no population-based parameter we investigated has power exceeding that calculated for these individual-based parameters (compare range in variability for various parameters on Table 2). This result — that impacts on individual-based parameters are more likely to be detected than those on population parameters — previously has been suggested [1], but we know of no other data or analyses that explicitly addressed the issue.

Table 2

Relative Variation in Differences Between Control and Near Impact Sites

The standard deviation of differences is standardardized across the various parameters by dividing by the mean parameter value at the Near Impact site. Given is the mean variability (and range) for each type of parameter.

	Index Of Variability (S_Δ / I)	
	Mean	**Range**
Population-based Biological Parameters		
Population density:	0.69	0.26 - 2.04
Re-entry Rate:	0.57	0.43 - 0.73
Emergence Rate:	0.49	0.44 - 0.58
Individual-based Biological Parameters	0.14	0.04 - 0.25
Physical and Chemical Parameters	0.12	0.05 - 0.19

Because of low levels of variability for physical and chemical measures (Table 2), we have similar, relatively high power to detect changes in these parameters as we have for individual-based parameters. For each of 4 physical/chemical parameters analyzed thus far, we again have greater power to detect a given size impact than for any of the population-based biological parameters

There are at least two explanations for our result that power is greater for impacts on physical/chemical parameters than on biological parameters, and that within biological parameters, individual-based measures have greater power than population-based parameters. In our analyses, power is high when the variability of the difference between control and impact sites over time, S_Δ, is low (Table 2, Figure 4). This variability is a function of two underlying sources of variation: within-site sampling error (variability among samples taken from the same site on the same date), and Site x Time interactions (see [25] for a discussion of optimal allocation of resources in BACIP). First, within-site sampling error is probably lower for physical/chemical and individual-based biological parameters than for population parameters because the latter are less efficiently sampled with a given level of effort. Second, physical/chemical parameters might be influenced more by large-scale oceanographic processes (and therefore will show a high degree of synchrony in fluctuations) than are biological parameters. In turn, biological parameters may be more sensitive to local conditions, reducing the degree to which values for different sites track each other through time. We currently are exploring this question by partitioning observed variance to determine the relative contributions of these two sources of error to each parameter type.

The dilemma posed by our results is that tests of impacts on the parameters of greatest interest to resource managers — population densities — have the least power for a given level of effort. One manner by which power can be increased is by increasing the number of sampling dates (true replicates). Figure 4 illustrates how power varies with the number of sampling dates and with variability. For a moderate amount of variability (0.25), increasing the number of sampling dates from 6 to 20 increases the power to detect a 30% reduction in parameter value at the impact site (relative to control) from about 0.2 to > 0.7 (Fig. 4). Unlike many factors that influence power, the number of sample surveys made is under the control of the investigator. However, it is critical that independence be maintained, and this may constrain how frequently sites can be sampled [12].

IMPLICATIONS

Throughout this paper, we have attempted to highlight problems and limitations associated with commonly used environmental impact assessment designs. It is important that such limitations be understood so that better and more effective assessment strategies can be developed and implemented. It is also critical that scientists, policy makers and regulators understand the limitations of each design to better interpret data that arise from each. We began the paper by stressing that many commonly used assessment designs often can lead to erroneous conclusions. These revelations are not new. Indeed they are well appreciated by many members of the scientific community.

However, some of the more subtle distinctions, such as between "Optimal Impact Assessment" and BACIP designs, are not widely appreciated by regulators or organizations conducting assessment studies, with the consequence that flawed designs still are commonly used (e.g., [21]). As a result, most attempts to provide the most rigorous scientific information concerning effects of a localized perturbation fail (for a still relevant review, see [3]). The practice of collecting equivocal data using inadequate assessment designs serves little interest, is unquestionably wasteful, and fails to ensure that the project or development in question is environmentally sound.

Although not widely utilized, the BACIP assessment design [12] has been employed successfully in a comprehensive study of the ecological impacts on the marine environment from the operation of a coastal power generating station [9]. BACIP avoids many of the interpretation errors associated with more limited designs. As such, BACIP is one of the most powerful (and therefore preferred) designs for the assessment of localized environmental impacts from point-source disturbances. This entails the explicit recognition that a time series of "baseline" data is needed before the commencement of the perturbation. Further, our results indicate that BACIP may lack sufficient statistical power to detect many impacts on parameters of most interest to regulators (e.g., population densities). Power of a BACIP test can be improved by increasing the number of sampling surveys (i.e., true replicates), and by increasing the number of samples taken at a site within a survey (i.e., the precision of each replicate). Increasing the number of samples (within a survey) will increase power if a large part of the variation in Control-Impact differences is due to sampling error. On the other hand, increasing the number of surveys will be helpful in most situations due to the influence of natural temporal variation in the Control-Impact differences. However, while a large number of surveys might be necessary, surveys must be spaced sufficiently in time to ensure independence. Thus, the application of BACIP requires extensive planning and foresight. To do so may require a fundamental change in the regulatory process. Regulators and policy makers must allow for a sufficient period of study prior to the perturbation if the goal is to obtain rigorous scientific evidence concerning localized effects.

Of course it will not be possible to conduct an appropriate BACIP assessment study for every new point-source development. Whenever a BACIP approach is employed, additional research should be undertaken to generalize results and thereby provide insight into other situations. This can be accomplished by examining the mechanisms by which environmental perturbations affect marine resources. Indeed, the resolution of environmental impacts ultimately requires this level of comprehension, and mechanistic approaches should be an integral part of any assessment study (be it a BACIP design or not). We need to understand the processes by which changes in the chemical and physical attributes of the environment alter the physiology of individuals (e.g., metabolic rates, energy allocation), how this altered physiology influences vital rates (e.g., birth, death, migration and growth rates), and finally, how these altered vital rates influence population characteristics (e.g., age-structure, density, production). This approach requires mechanistic studies at the toxicological, developmental, physiological and ecological levels, and which are integrated via dynamic (mathematical) models that are rigorously tested under field conditions. This will lead to better understanding of underlying processes, and thereby enhance our ability to predict ecological effects.

There is another compelling reason for an emphasis on mechanistic studies, either in concert with a BACIP assessment or as a "stand-alone" approach. Regulators and resource managers ultimately are interested in protecting marine resources from adverse impacts. The ability to mitigate or ameliorate adverse ecological effects will be greatly strengthened by knowing which attribute(s) of the perturbation are responsible, and how the effect(s) are generated. These issues can be addressed only through the type of mechanistic studies discussed above; environmental assessment designs such as BACIP only can provide information on the existence (and magnitude) of effects, and cannot address the underlying causes. Although environmental agencies have historically been hesitant to fund such "basic" research, there now seems to be a growing appreciation that resolution of critical environmental problems can only be achieved through rigorous development and integration of basic scientific tools within an applied context.

ACKNOWLEDGMENTS

We thank Jim Bence, Joe Connell and Allan Stewart-Oaten for helpful discussions and comments. Many people, particularly Don Canestro, provided valuable assistance in the field and laboratory; Bonnie Williamson rendered technical assistance. Ed DeMartini graciously supplied the data on pink surfperch, which we present with the permission of the Marine Review Committee. This research was supported in part by the University of California Coastal Toxicology Program and the Minerals Management Service, U.S. Dept. of the Interior, under MMS Agreement No. 14-35-0001-30471 (The Southern California Educational Initiative). The views and conclusions contained in this document are those of the authors and should not be interpreted as necessarily representing the official policies, either express or implied, of the U.S. Government.

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Seabirds as Indicators of the Oceanic Environment

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Abstract. Oceanic ecosystems are difficult to study. Even with the most advanced equipment, measurements of fish and invertebrate populations are hard to make because of their temporal and geo graphic patchiness. Additionally, changes in patterns of abiotic factors can affect the entire food web, and these changes often alter both abundance and distribution of important commercial species as well as species on which many other members of the food web depend. Two major questions that managers need to answer are: can these changes be predicted, and what is an indicator measure ment of these changes?

The importance of seabirds to humans, besides their aesthetic or food values, is that they are among the best biological indicator species of the health of the ocean. They are colonial; therefore they are visible. They return to the same sites each year to nest, and in this way are not geographically or temporally patchy as are other oceanic organisms. At seabird colonies, long-term studies can easily be conducted. Annual numbers of breeding birds, and measures of reproductive success such as laying, hatch ing, or fledging success, or chick growth are good indices of population trends and stability as well as those of their prey. Other indicator measurements are: weights of adults, of eggs, and fledglings, or species composition of the colony. Seabird populations can be measured in this way to predict events such as an El Niño or a collapse of a fishery. With a greater investment of time, various measurements of seabird prey can be made, and the subsequent charting of prey numbers, frequency and weight will give a good indication which changes are occurring in the prey base and thus in the surrounding oceanic environment.

The quest for understanding the interactions among abiotic and biotic factors in the oceanic ecosystem has been ongoing since humans started thinking about the sea. Today, many thousands of dollars are spent on monitoring oceanic parameters and although much is now known, much is not. Unlike terrestial ecosystems, the oceans remain hidden from easy study. Indeed, until this century, study and analysis of much of the ocean remained rudimentary at best. The measure of abiotic factors seemed to be the simplest data to obtain. Measurements of sea temperature and salinity, as well as above-ocean measures like rainfall and air temperature helped to provide details of the stability and change of physical factors over a period of time. However interpretation of these data on more than a local scale was lacking.

With the arrival of electronics, the study of other abiotic parameters like currents, tides, and deep-ocean temperatures were possible. With the advent of satellites, global assimilation of data gathered from techniques such as photo-interpretation of data like ocean temperatures became achievable (e.g. 1). Events like El Niño and trends like global warming were able to be detected more accurately and rapidly via state of the art computer modeling (2).

The measurement of abiotic factors in the ocean is important because these factors can influence biota from plankton to whales (e.g. 3, 4, 5, 6, 7, 1, 8, 9, 10). Likewise, distribution and abundance of prey populations influence distribution and abundance of other marine organisms. Abiotic factors often covary with distribution of animals, yet there is rarely geographic or temporal stability of these factors. From a practical standpoint, the identification and measurement of critical levels of abiotic or biotic factors that may control population numbers of certain commercial species becomes important for management of these populations.

Oceanic biota are inherently difficult to sample because the ocean is a fluid medium. They are not only patchy, they move, and are therefore unpredictable. A fish school present in the morning probably will not be in the same area two hours later. Finding the same school again is a much more difficult problem than locating a flock of birds in a deciduous forest. Humans are terrestrial beings and our senses are most tuned into terrestrial cues.

The oceanic sampling of biotic parameters probably started out as simple counts. Recently, more advanced techniques such as aerial detection of fish schools, analysis of water samples for grams of carbon per cc, or bioaccoustics, where sonic pictures enable researchers to follow movements of fish, gave researchers a more complete picture of the oceanic ecosystem.

Many of the techniques developed for following fish schools were introduced because the species for which they were developed were of commericial value. These techniques were often later refined and used for scientific applications such as analysis of predator- prey interactions or for correlations between distribution of phytoplankton or invertebrate prey and seabird populations (11, 7, 12, 13, 14, 15). Yet these measures still depended on first finding, then measuring the concentrations of plankton or fish. Sampling was still a problem. Even with the use of electronics to find oceanic creatures, we may either wrongly sample them or miss them altogether.

Certain cues can be used to locate biota. Commercial fishermen have tried to solve the dilemma of finding what they want to catch, by sighting on a predator species, usually marine mammals, which hunt the same prey they seek. Marine mammals, unlike the majority of fish, spend a comparatively large amount of time at the surface of the water, and are thus visible to humans. For example, some commercial fishermen sight on dolphins to find yellowfin tuna, a common prey of dolphins. Sometimes fishermen also look for flocks of seabirds to find swarms of krill. Timing, however, is still a problem, for these predator species also must be located, and they too are patchy and unpredictable.

If we continue to sample fish or invertebrate populations the way we do at present, we may not detect early deleterious or critical decreases in their populations. Subtle incremental changes may be additive and we might have no way of knowing whether or not we should be alarmed at these changes until perhaps the populations have already reached critically low levels. We need to be able to anticipate changes in the ecosystem and to predict what effect these changes will have: from the smallest prey to the largest commercially desirable species. We need early warnings.

Three recent events that might have been predicted but were not are: 1) the collapse of the herring fishery off of Norway in the 1960's (16), 2) the collapse of the anchovy population off Mexico in 1990 (D. Brewer, pers. comm), and 3) the radical decrease of the pollock fishery in Alaska to 25% of its normal levels (17). Could these collapses have been foreseen? Were there any predictors of these potential collapses that could have been monitored instead of the fish populations themselves, which clearly did not predict their own demise?

One way to get at the problem of predictability is to understand that changes in the food web are often magnified through higher trophic levels. Even small changes in the prey populations will be reflected in concomitant changes in the associated predator populations. Thus, repeated counts of large predators may be a good indicator of biotic and abiotic changes in the ocean. Good predators to measure would be top carnivores like salmon and tuna that feed on common species like capelin, sandlance, herring, smelt and anchovies, on which many other species feed. The problem in prediction here is that these fish carnivores are difficult to locate because for the most part they remain invisible to researchers. Even using feeding assemblages of marine mammals or birds to locate these fish does not help because the prey themselves are still unpredictable in place and time. Clearly, something that is stable, preferably stationary and present year after year, as well as being easy to measure needs to be found.

Ocean transects are stationary and repeatable and have been used to sample everything from physical and chemical parameters to seabirds and mammals. If enough of an area is covered on a regular basis, ocean transects are a good way to monitor long-term changes and variations (14). However they are expensive and thus are often not conducted regularly throughout the year over a large enough area and often not over a long enough period of time. Infrequent or incomplete transect sampling probably does not give accurate information on natural variation and abundance and certainly does not yield good predictive values. The biota found in these transects are transient and the transects only provide a snapshot of what is present. Data gleaned from them are useful to correlate carbon or temperature or salinity with population abundance of predators or prey species (5, 7, 1, 8 , 14), but not to count the same fish population from month to month.

However, there are some marine-associated species that are not only visible but are also predictable and stable in place in time. These are seabirds breeding at colonies. Seabirds return to nest, usually annually, to their natal colonies. Their populations frequently do not change drastically over a number of years. Since seabirds are long-lived, any increases or decreases in the overall colony numbers will probably not reflect a short-term output or decrease of chicks during a single breeding season. Seabirds are wide-ranging upper trophic level consumers and their colonies are often found near areas of high oceanic productivity (18). They are visible and easy to monitor because they are stable in space and in time.

Abiotic and biotic changes in the oceanic environment will often be reflected in certain changes at seabird colonies. Since many seabirds are also top predators, they are strongly affected by changes not only in their prey base but also in prey species a few trophic levels below them. Additionally, their populations are affected by changes in abiotic factors which influence all aspects of their food web (19, 20, 21, 10, 3, 22, 23, 43, 24, 25, 11, 26, 27).

The simplest but not necessarily the most accurate method of tracking seabird populations and determining if subtle changes are affecting them is to census them on their colonies over a long period of time (28, 29, 30, 31, 32, 33, 34). This is easy and fairly inexpensive. They should be censused during the peak of the breeding season (early-to-mid nesting) when the largest number of birds are present on eggs (see 35). At least one but prefera bly several censuses should be made by several observers to increase the accuracy of counts, since nesting can be advanced or delayed by both biotic and abiotic factors (36, 37, 38). Annual counts should be consistent in a well-established colony. An oil spill or collapse of a major food stock or radical change in ocean temperatures will affect numbers quickly (e.g. 34, 16, 3, 39, 32, 40).

To predict when and what to census, the natural history of the species needs to be known. Black-legged Kittiwakes, for example, will sit on nests even if no egg is present to retain ownership of their scarce nesting site (36). Tufted Puffins often stand outside their burrows in the early morning, where at other times they are inside their burrows and are thus invisible for censusing. Pigeon Guillemots are best censused at both high tide and during the morning to obtain optimal colony attendance (41).

Many researchers have looked at population numbers over a period of years (28, 23, 31, 33, 34, 18). In these studies, the long- term effects of abiotic and biotic factors can be separated from short-term effects.

Another indicator of abiotic and biotic conditions is the reproductive success of marine birds (42, 15, 43, 40, 26, 44). There is growing evidence that abiotic factors like ocean temperatures, rainfall, and wind can influence abundance, distribution, and catchability of prey in some oceanic areas (45, 46, 47, 48, 3, 4, 21, 10, 49, 6, 50, 51, 52). Prey abundance and availability in turn influence reproductive success (53, 42, 3, 21, 10, 58, 50, 51, 43).

Thus, a better measure than population surveys of changes in abiotic or biotic parameters is reproductive output of seabird colonies over a long period of time (28, 40, 54, 55, 3, 43, 11, 44). Short-term fluctuations in reproductive output are the norm, with many of these long-lived species sacrificing egg or chick success in a scarce food year (e.g. 53, 55, 3, 15, 56, 23, 57, 12, 10, 58, 59). Yet, if studies are conducted over a period of years, any significant changes in output will be detected. Drastic changes in success from one year to the next could most likely be a reflection of concomitant changes in the prey base. This could mean either that prey are unavailable (have moved elsewhere due to abiotic factors, e.g. El Niño's bringing in warmer waters (60, 61, 55, 3) or are stratified at depths where they are uncatchable (10), or else that prey stocks are low. There are many examples of this (3, 21, 10, 36, 62, 63). Most recently, just before the anchovy collapse off Mexico, there was a breeding failure of Brown Pelicans that nest on the Cornado Islands close to the anchovy fishing grounds (D. Brewer, pers. comm.). Up to 80% of these nests were abandoned in March before the summer anchovy collapse. Brown Pelicans could have been used as a predictor of a decrease in an important commercial species, and perhaps if commercial harvesting of anchovy had been stopped at that point, the collapse may not have occurred.

The major problem with counting offspring is knowing at which stage to census them: at laying (e.g. number of eggs laid), at hatching (e.g number of eggs hatched), or at fledging (e.g. number of chicks fledged). There are accuracy problems with each stage.

If only eggs are counted, this census will reflect only abiotic and biotic events affecting the parents up till egg laying. Abundance of food is known to affect laying either by increasing the number of eggs or by increasing the eggs' weight (64, 65, 37, 26, 66, 10, 58). Thus egg weight should also be measured. Caution must be used if the species migrates back and forth between the breeding and wintering grounds, for egg counts or weights may reflect the conditions present on the wintering grounds. In addition, the census date is important to ensure accuracy (35). A knowledge of the chronology of the colony under study is thus needed.

Another cause of reproductive failure could be the presence of a pollutant, such as DDT (67) or petroleum (68, 69, 70, 71, 72, 73, 29, 34) which could affect the birds either directly (e.g. eggshell thinning) or indirectly (negative effect on the food web).

Weather and prey may also drastically change over the next stages, the incubating and chick stages, and thus the egg or chick success is also important to measure during these periods. The percentage of eggs hatched gives an indication not only of abiotic and biotic factors occurring during incubation or hatching, but also may give an indication of the presence of pollutant loading if hatching success is low but laying success is high (67).

If the population is regulated more by density independent factors like weather or denisity dependent factors like food availability and not by nest space, then the number of offspring that are produced in a colony may be the most accurate measure of population stability (81). However sometimes this is the most difficult stage for which to obtain an accurate count. Many chicks hide in vegetation away from the nest until they fledge, (49, 36, 38) and other chicks leave the colony before they can fly (e.g. Ancient Murrelets, murres) and the researcher has difficulty estimating survival of chicks who fledge at sea (36, 74). Weight gain by chicks has also been used as an indiator of marine resources in oceanic systems (47, 75, 64, 76, 3, 26).

If numbers of offspring produced are low, the next step is to try to determine the cause. If abiotic factors have been measured concurrently, corresponding changes may be found. If a correlation is found, we need to know if the abiotic parameter is of a direct or indirect nature. For example, does an increase in precipitation affect chick survival directly by soaking the feathers, or does it drive prey more deeply in the water where they are inaccessible? Do changes in temperature chill or overheat the eggs or chicks or do they drive the prey into other areas (3, 40, 10)? Might rain in the previous season affect chick output the next season by diluting the ocean surface where fish eggs (next year's prey) are laid inshore? On the other hand, might a large amount of precipitation wash fish eggs out to sea or kill them by the change in osmotic pressure? These factors all need to be determined.

If the effect of the abiotic factors can be established to be direct, then is this change just a brief deflection or does it herald a long-term overall change in oceanic conditions? A rain storm or cool temperatures during hatching may rapidly decrease the chick output of any given year. However, unless this abiotic pattern is repeated year after year, and seems to be an established change, then effects on the population as a whole will probably be small, barring other factors. Only measurements on long-term studies will provide an answer. Presence of DDT and lack of success of Brown Pelicans is probably the best known study on direct cause and effect (67). After DDT was banned in the United States, the success of Brown Pelicans there increased remarkably after having been on a decline since use of DDT began.

To obtain the best overall picture of what is occurring throughout various trophic levels in the oceanic ecosystem, a varitey of seabird species should be monitored. If several species at a colony are studied, each one may be an indicator of a different marine resource, depending on its prey preference and method of foraging. Thus the condition of many marine resources can be observed independently through sampling their seabird predators (42, 26, 9).

The determination of abundance and distribution of prey over a period of years is one of the most accurate means of sampling the long-term effects of oceanic parameters on all biota in the food web. Sampling for invertebrate or vertebrate prey directly is difficult, because prey are patchy. The best way to analyze this problem is to use seabirds as biological sampling devices. If quantitative data on fish populations are required to make predictions about their population trends, seabirds can also be used as the samplers. Seabirds are upper trophic level consumers, wide-ranging and diverse in their feeding habits. They are probably the best samplers of what available fish and invertebrate populations are present (77, 23, 24, 43, 11). By obtaining prey samples from seabirds, the presence of some abiotic factors, e.g pollutants, can be observed (79, 67, 78).

One method to sample prey would be to search for flocks of feeding seabirds and collect an adequate number of birds for analysis. Although seabirds are more visible than fish, they are still patchy and fairly unpredictable, and the problem of obtaining samples that reflect oceanic changes may still not be solved because of poor sampling techniques. Likewise, stomachs are often empty from seabirds collected in this way (P. Baird, G. Sanger, unpubl. data). The best way to sample prey is to obtain food from adults or chicks in the seabird colonies (36, 80). This way, information on seabirds' prey may be added to data on their populaton numbers and reproductive output. Thus one can obtain a more complete picture of what is occurring in the oceanic ecosystem.

Concurrent studies on prey and predators have been carried out on a variety of species including the Elegant Tern, Brown Pelican, Rhinoceros Auklet, Black-legged Kittiwake, Common Puffin, Glaucous-winged Gull, Tufted Puffin, and California Least Tern (81, 82, 11, 12, 63, 10, 58). The main drawback of analyzing prey is that it is more time consuming than censusing adults, eggs, or chicks. This defect, however, is overridden by the more accurate picture given by prey analysis, especially if concurrent measurements on abiotic factors have been made. Conclusions might then be reached of how the measured abiotic factors might be affecting both the prey base and the seabirds, (e.g. 10, 58). Likewise, care must be taken to measure the same age class and breeding stage of bird each year, for it has been shown that different age classes and different reproductive stages consume a different suite of prey species (59).

The natural history of the most important prey species of a seabird population is necessary to make inferences about how these prey might be affected by abiotic changes in the environment (18). Do they school? Do they stratify at depth at certain temperatures? Do they lay their eggs inshore where they might be heavily affected by runoff from an unusually wet year?

Common measurements of prey are frequency of occurrence, percent numbers, weight and length and overlap (77, 83). The natural history of seabirds must also be taken into account when analyzing prey data because it is important to know whether the birds are feeding over a large segment of the water column or if they forage only on the surface. Surface feeders are often more limited in their choice of prey than are seabirds that forage throughout the water column.

Whether a seabird is a specialist or a generalist must also be accounted for when analyzing prey from year to year. If a seabird is a generalist, there are alternate foods to choose from if a favored prey is absent one season. These species can turn to other prey to feed chicks and to raise a healthy crop of young. On the other hand, if the species is a specialist, then a crash in a particularly important prey will severely affect their output of young (e.g. 58).

Even for a generalist, a change in diet breadth is an indication that something has happened to the preferred prey (36). It is therefore best to study more than one species of seabird to obtain the best picture of which changes are occurring in the ecosystem. And it must be kept in mind that for the same effect, e.g. food shortage, different species react in different ways (42, 26, 9, 10).

If we monitor more than one species, say a generalist and a specialist, a surface feeder and a diver, a surface nester and a burrower, then we can narrow down the factors which might be affecting the reproductive output, population numbers or food. A year with a high rainfall may not only soak newly hatched chicks, it might also push the preferred prey into deeper waters where only divers are able to catch them (10). A typical finding in this situation might be: presence of preferred species in divers and not in surface feeders, expansion of the prey base by generalists, higher relative reproductive output by burrowers than by surface nesters, a low reproductive output by surface feeding specialists (10), or low adult weights and smaller clutches and even a change in numbers of extra-pair copulations in other species (58).

CONCLUSION

Seabirds, because of their highly visible and relatively stable colonies, can be used as indicators of the health of the ocean. Detection of changes in populations of fish or invertebrates is difficult because of their lack of visibility and their patchiness in space and time.

Observations of annual population numbers of seabirds, of their annual reproductive success, and of their prey can reveal patterns which are often correlated with food-rich or food-poor years, (53, 42, 55, 3, 21, 10, 63, 51, 56, 43, 57, 74). Prey abundance in turn can be correlated with abiotic changes in the oceanic environment (47, 48, 4, 50, 51, 52, 6, 10, 58). From these observations, predictions can often be made about trends in fish or invertebrate populations or even in long-term changes in abiotic factors. Thus much of the information about the oceanic ecosystem that appears hidden or obscured or at best difficult to obtain, can actually be found when seabirds are used as biological indicators.

ACKNOWLEDGEMENTS

The idea for this paper was first hatched in the summer of 1977 on a seabird colony off Kodiak Island, Alaska. These fledgling ideas were subsequently presented at a meeting of the Pacific Seabird Group in December 1977, and much discussion ensued with especially: Gerry Sanger, Alan Moe, and George Hunt. I thank numerous others who helped me refine my ideas for subsequent presentation at the Southern California Academy of Sciences meetings in 1991. I especially thank Dee Boersma and Kees Vermeer, who contributed excellent suggestions for improving a first draft of the manuscript. I also thank Sue Yoder, University of Southern California Sea Grant Program, for her forebearing and relentless encouragement and enthusiasm.

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What Can Southern California Learn From The EXXON VALDEZ Oil Spill Experience?

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Abstract. The EXXON VALDEZ oil spill is the largest to have occurred in the United States. The accompanying shoreline treatment program was the most extensive and expensive conducted for any oil spill to date. Examining the experience and events surrounding the spill and its cleanup from a biologist's viewpoint suggests several lessons on oil spill study design, policy, and planning that are relevant to southern California. Examination of spill scenarios from the EXXON VALDEZ and Huntington Beach suggests that the economic value, benefit, and environmental cost of cleanup vary substantially according to the nature of the biota, resource values, and past history at the spill site. This analysis also suggests that, in developing oil spill policies and contingency plans, we need to recognize the lessons of history and answer important questions concerning: 1) resources at risk; 2) intentions regarding cleanup and litigation after a spill; 3) real environmental and economic costs of cleanup; 4) the cost-benefit ratios of cleanup and litigation; and 5) adequate funding for valid scientific studies prior to a spill in areas at risk. Important data gaps include: 1) baseline data for important areas and resources; 2) understanding of chronic effects on communities; and 3) information on effects and effectiveness of cleanup and bioremediation.

INTRODUCTION

The disaster surrounding the EXXON VALDEZ oil spill is a case history of lost opportunities. There were numerous opportunities to avoid the accident. The weather provided considerable opportunity to effect containment of the spilled oil, although the sheer volume of oil probably would have ultimately precluded effective containment. The event provided an excellent opportunity for careful studies in a subarctic system of the effects of catastrophic contamination by crude oil on marine birds and waterfowl, marine mammals, major fisheries, as well as the plethora of organisms that play the supporting roles in the lives of these more prominent natural resources. Finally, the spill provided an excellent opportunity to study and evaluate a broad range of cleanup and treatment alternatives. However, many of the opportunities were lost because of poor planning before the spill; poor implementation of good planning; the confusion surrounding the spill; the inexperience, ill will, or divergent motives of many of the participants; and, finally, the litigative environment that the event engendered.

In my view, the handling of the spill response and evaluation was fraught with problems from the beginning and the entire process is in need of many changes. However, the EXXON VALDEZ experience provides many excellent lessons that can help the State of California to cope with the effects of major oil spill events. Specifically, the spill brought to light historic, environmental, regulatory, treatment, and scientific lessons. In this paper, I will address some points to be learned from each of these lessons, within the context of the EXXON VALDEZ experience and other notable events that assist in providing a proper perspective for each lesson.

While the lessons are clearly the focus of this paper, it is useful to preface that discussion with an outline of my own experience on a few of the numerous studies that have been undertaken by the general scientific community in the wake of the EXXON VALDEZ oil spill. The paper concludes by identifying: 1) several significant data gaps that will prove to be of concern in conducting accurate analyses of future spills; 2) several issues in need of further discussion; and 3) the major conclusions reached during the course of my research.

Along with several of my colleagues, I was involved with studies funded by Exxon and the National Oceanic and Atmospheric Administration (NOAA) following the spill. Outlined here are some of the studies that I have been personally involved with, for it was observations made during the course of these investigations that formed the basis for the discussion included in the lessons that follow.

OVERVIEW OF EXXON NEARSHORE STUDIES

Baseline Response. The initial tasks in the Exxon-sponsored studies focused on obtaining baseline data from sites that had not been contaminated by the moving mass of crude oil. We visited sites on Knight, Montague, Evans, and Bainbridge Islands, as well as on the mainland on the western side of the sound, working around the fast-moving slick in order to sample at locations that had not yet been oiled (Figure 1). Based upon a study design devised during a previous NOAA study [1], we concentrated on sampling in protected or sheltered locations because of the higher productivity and sensitivity of biological assemblages in those habitats and the higher probability that oil would persist there in the absence of strong wave action or currents. Oil was on the water within two miles of all twelve initial sampling sites and we felt confident that many of the sites would be oiled subsequent to our visit. In fact, however, the winds and currents moved the slick such that only one site (Bay of Isles on Knight Island) was oiled after we sampled it and that location was only lightly contaminated.

Late in the first survey (early April), Exxon asked us to survey Outside Bay on Naked Island before the EXXON VALDEZ was moved there for evaluation and repairs, and then to examine several oiled sites. For the oiled sites, we selected a gradient of oiling intensity ranging from lightly oiled (previously surveyed Bay of Isles) to very heavily oiled (Northwest Bay on Eleanor Island). A few weeks later, we returned to resurvey some of the initial sites and establish some new sites.

During the initial "baseline" surveys, we conducted a multi-faceted biological and chemical survey at each site. We sampled phytoplankton, pelagic and epibenthic zooplankton, demersal fishes, subtidal and intertidal epibenthic and infaunal assemblages, and PAH concentrations in the water, sediment, and tissues of dominant invertebrates and fishes. Moreover, we collected samples of several dominant inver-

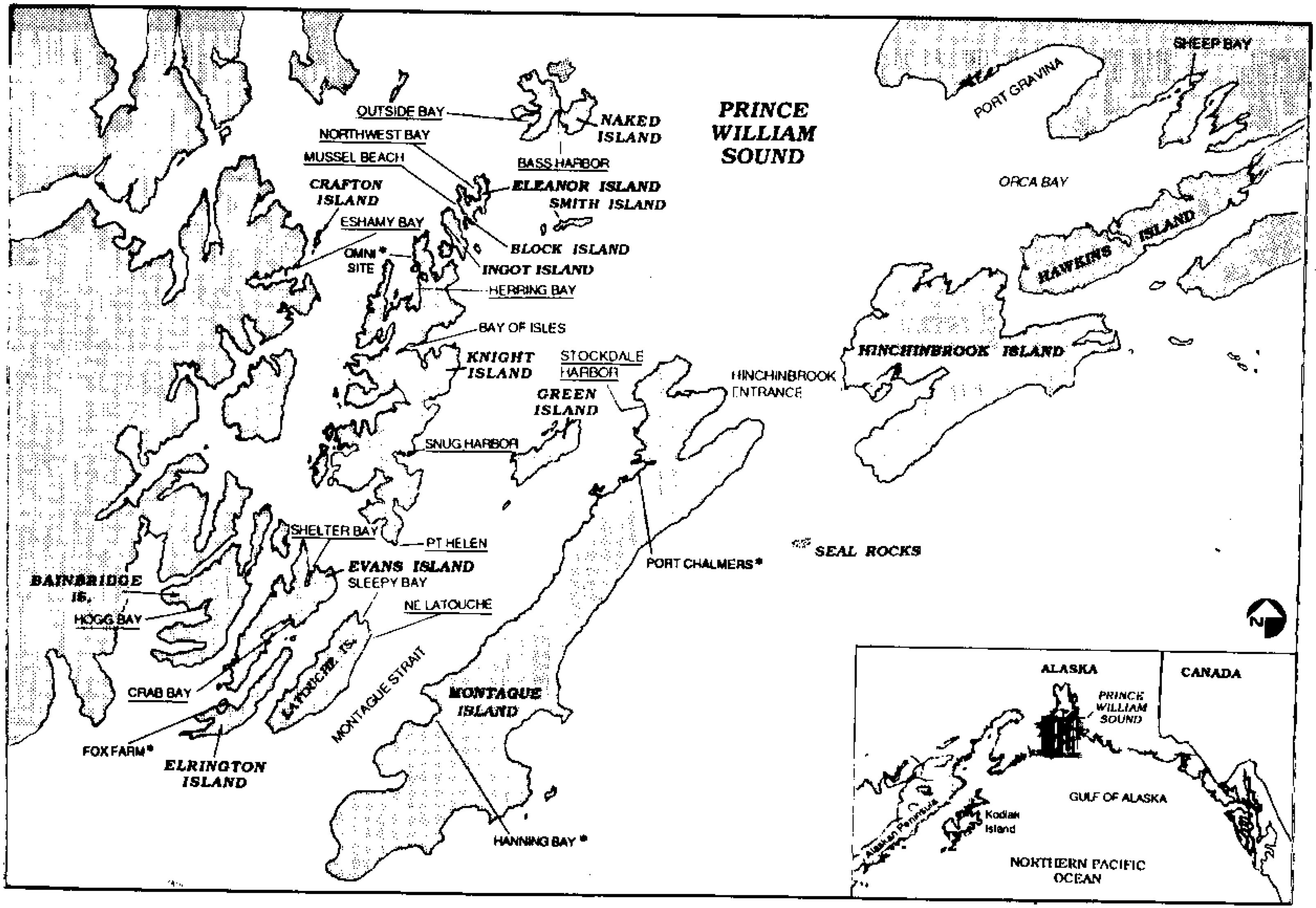

Figure 1. Prince Williams Sound study area are sample locations for Ogden's NOAA and Exxon studies. The underlined locations were both Exxon sites and NOAA study sites. The locations note with * are Exxon sites only, and plain text locations are NOAA study sites only.

tebrates to permit examination of population dynamics at a later date. Subsequently, we initiated studies of kelp growth rates and eelgrass beds to assess the impacts of hydrocarbon contamination on these valuable resources.

Like many in southern California, my understanding of the biological effects of oil was based primarily on examination of literature and various summary documents for several major spills, including the blowout that had occurred in the Santa Barbara channel in 1969. I had some definite preconceptions and expectations on the severity of effects. As a consequence, I expected to observe "mass destruction" as our examination of the effects of the oil spill continued. Therefore, I was quite surprised when, by late June, while we had observed substantial damage, the effects on the intertidal biota were not catastrophic. Mortality was observed in many species, for example limpets, chitons, and mussels, but generally, we did not observe the mass mortalities that we expected or that had been predicted.

Recovery Studies. By late June, about three months after the date of the spill, most of the original heavy crude oil on the surface of the water had washed out of the sound or had been recovered and any further contamination was caused by reoiling following resuspension of grounded oil. Moreover, it appeared that mortality caused by initial massive contamination (either from toxicity or smothering) was no longer a concern and that further effects, either lethal or sublethal, would be caused mainly by chronic contamination. As a consequence, we commenced an evaluation of the recovery of the assemblages at oiled sites and adopted several new sites in order to provide a wider variety of habitats to expand and fill in the sampling matrix.

Treatment Effects Assessments. Toward mid-summer, after shoreline treatment operations had been in progress for about two months, Exxon responded to concerns voiced over the potential impacts of the cleanup by conducting a study on the effects of Omni-barge treatment, the most vigorous of the high-pressure, hot-water cleanup techniques. Following this study, a more general study of treatment effects was initiated. This work was not initiated until early September, when most heavily oiled sites had already been treated and only two weeks before treatment activities were terminated.

Meanwhile, serious cleanup operations commenced in mid to late May. We learned that cleanup operations and shoreline treatment are quite disruptive, especially in a remote, pristine environment like Prince William Sound.

By the end of 1989, Exxon's focus had changed from one of assessing the effects of the spill and the treatment to one of defense against the many suits being brought against Exxon, some in conjunction with the Natural Resource Damage Assessment (NRDA) program. Since we had little experience in the conduct of NRDA studies, we were released from our commitments to Exxon and began looking for other funding to continue our examination of effects and recovery.

As of this time, we are still constrained from discussing most of our 1989 data because these were collected under the auspices of Exxon. Data have been released for the Corexit 9580 M2 and the Omni-barge studies [2 and 3, respectively].

OVERVIEW OF NOAA STUDIES

National Oceanic & Atmospheric Administration Studies. We found that no funding was available through state or federal agencies for scientific studies of the effects of the spill or to document the recovery of the biota. However, we found that NOAA's Hazardous Materials Response Branch, the group responsible for providing scientific advice to the Federal On-Scene Coordinator (FOSC) on spills of contaminants, was interested in obtaining better information on the effects of shoreline treatment so they would be better prepared to advise the FOSC on the conduct of the cleanup. Consequently, NOAA partially funded a study to assess the effects of the more aggressive treatment methodology. Additional funds were solicited from the Minerals Management Service, the Environmental Protection Agency (EPA), the U.S. Coast Guard, the American Petroleum Institute and the Marine Spill Response Corporation, and Exxon agreed to provide logistical support for major elements of the study.

Our ongoing NOAA studies are structured somewhat like the work we conducted for Exxon except that the primary purpose is to examine the effects of high-pressure hot-water washing; quantifying recovery is a secondary objective [4]. We have continued to monitor sediment and tissue hydrocarbons at some of the same sites but have also established several new sites (Figure 1) and discontinued looking at hydrocarbons in the water column. The principal biological elements of the program include examination of species composition and other community characteristics of the intertidal epibiota and infauna, and population features (growth and mortality rates, size structure, and reproductive success) for intertidal populations of four molluscs (blue mussels - *Mytilus edulis*; Sitka periwinkles - *Littorina sitkana*; a drill - *Nucella lamellosa*; and littleneck clams - *Protothaca staminea*), and subtidal eelgrass *(Zostera marina)* [4].

HISTORIC LESSONS RELATED TO OIL SPILLS

The logical reasons for conducting a cleanup generally pertain to protection of biological, economic, recreational, or aesthetic resources, or a mixture of concerns for these resources. The concern for oil on the water, sand, or rock is not so great as is the concern for the uses of the water, sand, or rock by animals and plants, the importance of specific animals and plants to man, or the uses that man has in specific areas for the water, sand, or rock. We typically don't clean rocks or water for the sake of the rocks or water themselves, but because of the support they give to the biota, fisheries, tourism, etc.

James Mielke, a specialist in marine and earth sciences for the Science Policy Research Division, Congressional Research Service of the Library of Congress, published a report on the short- and long-term impacts of oil spills [5]. His report, a summary of the "life cycles of six major spills," compiles available information on the environmental and socioeconomic damage caused by each spill and compares the magnitude of these damages, as shown by long-term studies, with the popular perceptions of damage resulting from the media coverage of each event. The spills that he evaluated were the Santa Barbara and Ixtoc I blowouts and the ARCO MERCHANT, BURMAH AGATE, ALVENUS, and AMOCO CADIZ tanker spills, each of which received extensive media coverage.

He pointed out that historically, it has been unusual for more than 10 to 15 percent of the spilled oil to be recovered. Efforts on the EXXON VALDEZ spill reinforce this assertion. Over $2 billion have been spent to recover a small percentage of the oil spilled in Prince William Sound. In fact, more oil was burned in engines during cleanup activities than was spilled [6].

Mielke makes several important assertions. First, his review indicates that spills are fairly short-lived events, i.e., the effects and residence time of the oil in the nearshore environment generally last less than a decade. The major ecological damage occurs within the first few months of a spill event. Generally, the media coverage is shorter but provides intense descriptions of a major environmental catastrophe. Media descriptions define the public perceptions of the effects and these perceptions are typically longer-lived than the event or its impacts. *The Washington Post* reported that the judge determining bail for Capt. Joe Hazelwood in 1989 referred to the spill as the worst disaster since Hiroshima, based on his media-based impression of the spill. The Santa Barbara blowout provides a relevant example of the persistence of this phenomenon; while still viewed as a major catastrophe by the public in southern California, the mid- and long-term ecological effects of the blowout were negligible.

Mielke observed that short-term impacts on the biota may be devastating but, in the larger picture, the effects have not been shown to be significant. Spills have not had an appreciable effect on world populations of animals or plants. Generally, mortality from a spill is far less than mortality from the annual sport or commercial harvest. In the case of waterfowl, more than 300,000 ducks are killed each season on the eastern shore of Maryland alone, compared to the estimated 350,000 to 500,000 killed over a much larger area by the EXXON VALDEZ spill.

Mielke's review indicated that cleanups generally yield minimal benefits and in some cases result in significant delays in recovery. This summary implies that cleanups, in some cases, do produce salutary effects. One can conclude from this that the first question that needs to be addressed for any oil spill, preferably in the contingency planning stage, is whether or not a spill cleanup should be initiated. If the answer is affirmative, it should be determined: 1) under what conditions it is worthwhile, 2) which habitats should be cleaned and which excluded, and 3) which types of cleanup measures should be employed and which types excluded.

ENVIRONMENTAL LESSONS FROM THE EXXON VALDEZ SPILL

What can be learned from the activities surrounding the EXXON VALDEZ oil spill? In Prince William Sound, the water was skimmed because of concerns over oiling or reoiling the shoreline and contamination of fisheries stocks such as herring and salmon. Rocks or beaches at specific sites were cleaned initially because of the imminent influx of calving pinnipeds and breeding birds onto specific haulouts or rookeries or the outmigration of juvenile salmon and the return of adult salmon. In the mid-term, the concern was for fisheries and the aesthetic value of the shoreline and the way in which that translates into tourist dollars. It is interesting, however, that it is very difficult to discover the objectives of the cleanup; many publications discuss the decision-making process and the organization but the driving objectives for the cleanup response are not clearly enunciated [e.g., 7, 8, 9, and 10]. Lindstedt-Siva noted that the goals of spill response are usually not stated [11]. Moreover, she claimed that national goals have evolved from a primary goal of removal of all visible oil toward one of minimizing the ecological impacts of a spill and but are now swinging back to the former objective. She pointed out that the national goals must be enunciated clearly before any spill contingency planning is completed and made a strong case that the objective should be to minimize the ecological impacts of a spill.

Implications of Our NOAA Data. Our studies for NOAA [4] suggest several lessons for consideration in the approach to policy related to oil spills and ensuing cleanups. These lessons pertain to sampling design, effects of oil contamination and shoreline treatment in several habitats, resilience of populations and community recovery rates, and the effectiveness of existing containment and cleanup equipment and strategies.

Our findings suggest that the biological benefits and costs of shoreline treatment vary by habitat type and elevation (shoreline location). In Prince William Sound, it appears that the biological costs and benefits of treatment of upper elevations on exposed boulder/cobble were not great; the general value of the biological resource on this habitat is generally low relative to most other habitats and the tolerance and resilience of the biota in these habitats are great; this results from the combination of the selection processes of nature and the stressful nature of the environment in which these organisms live. Thus, while the biological cost of the treatment (and, incidentally, the damage from the hydrocarbon contamination itself) was low (relatively little biological damage), the benefit of the treatment to the biota was also low and, ultimately, performing the cleanup in these habitats was biologically neutral. The cost in terms of human resources expended was not low, however.

At the other end of the spectrum, shoreline treatment with hot water resulted in low benefits and high biological costs on protected rock (Figure 2) and mixed soft (Figures 3 and 4) substrates in Prince William Sound. The biological assemblages in these habitats are generally richer, more productive, and more sensitive to both the effects of the initial oiling and the treatment. Generally, our observations suggest that oil exerted an appreciable negative impact on these habitats, but that high-pressure, hot-water treatment exerted a significantly greater adverse effect. Moreover, patterns ob-

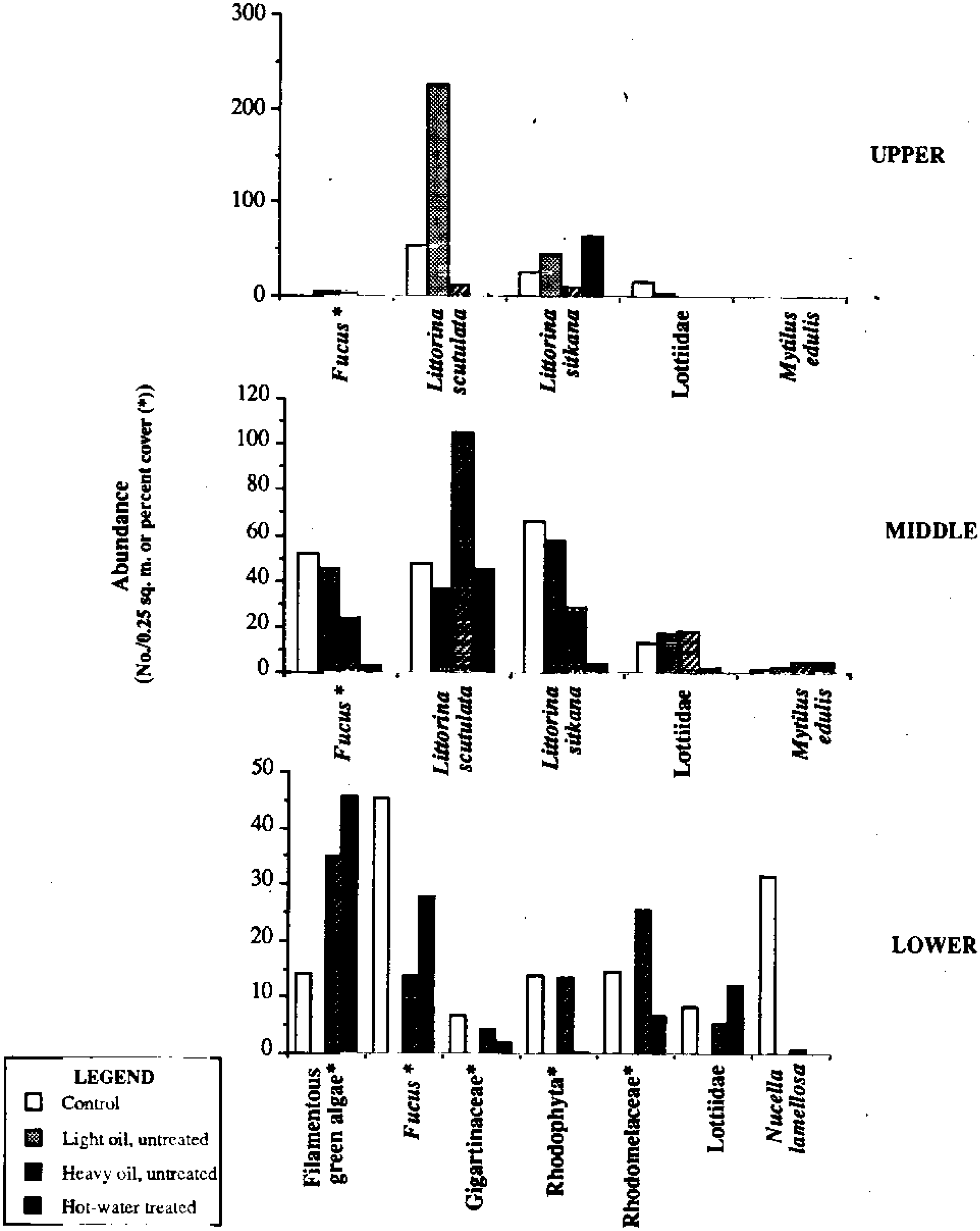

Figure 2. Abundance of major epibiota at protected rocky stations in Prince William Sound.

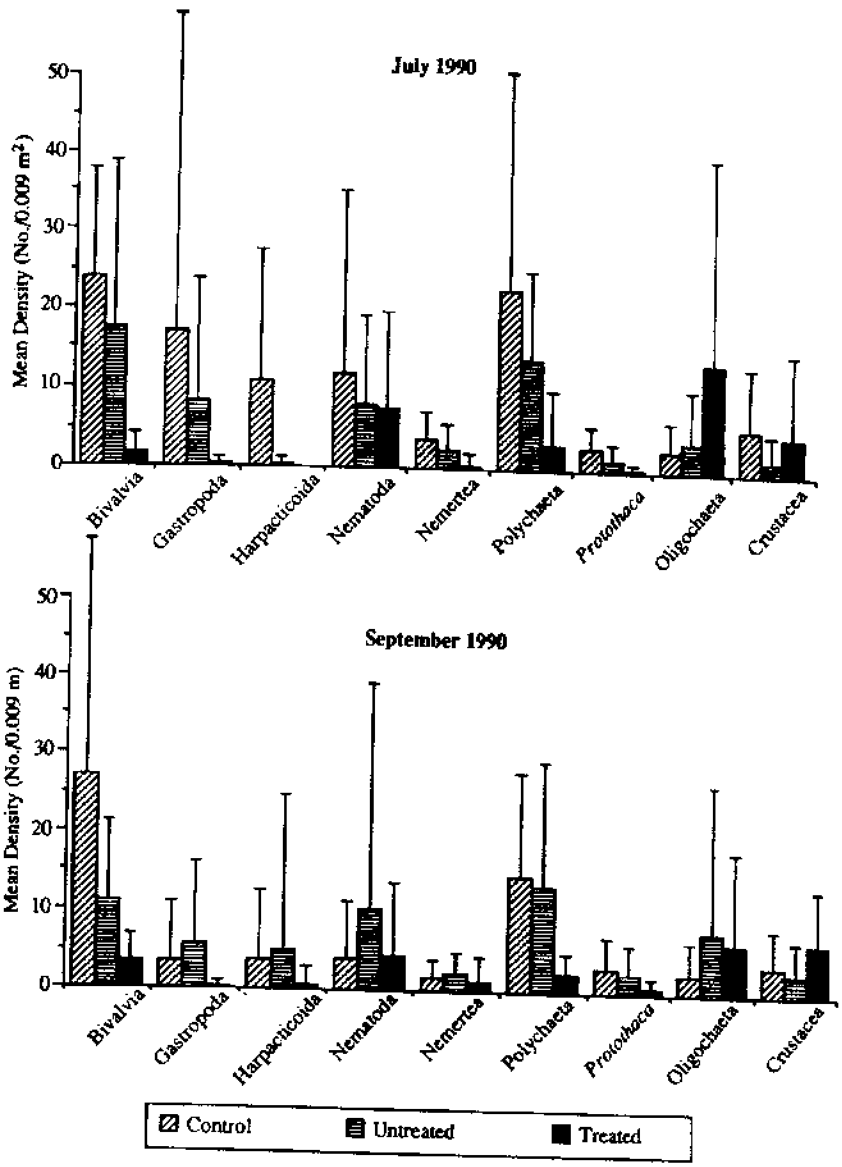

Figure 3. Mean densities (with standard errors) of major infaunal taxa at lower mixed-soft stations in Prince William Sound.

served to date suggest that recovery in treated areas is substantially slower than in areas that were not treated. In the case of clam populations (Figure 4), damage from the initial spill was modest in comparison with damage from the treatment, which was a sudden, catastrophic episode from which recovery will probably require decades.

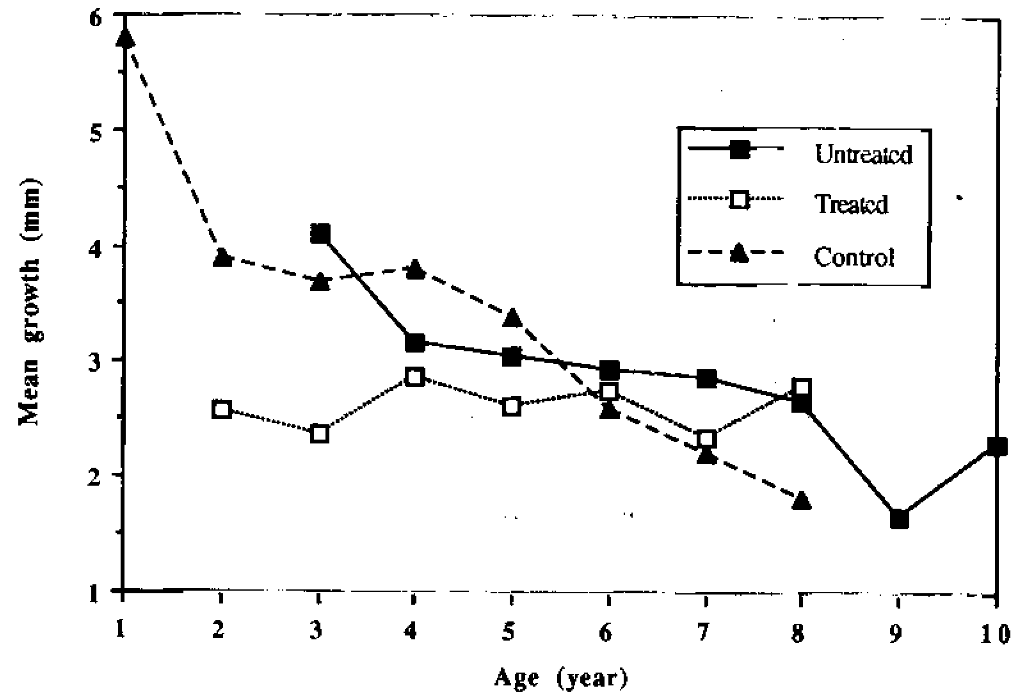

Figure 4. Clam growth at mixed-soft habitats in Prince William Sound

Moreover, it appears that an important effect of the cleanup using hot or cold hydraulic flushing was to move oil from the upper and mid levels of the intertidal zone, where its effect was restricted to relatively tolerant species such as barnacles and littorines, down into the lower intertidal and shallow subtidal regions, which are more productive and support more sensitive types of organisms (Table 1).

It appears that season, habitat, and treatment methods are important factors in determining biological benefits and costs of shoreline treatment. It was beneficial to cleanup the haulouts and rookeries prior to the return of the marine mammals and birds for calving, breeding, and rearing activities; however, at other times of the year, it is unlikely that a similar level of treatment would produce similarly salutary effects. By way of comparison, at the AMERICAN TRADER spill onto heavily used sand beaches in Huntington Beach, it appears that the cleanup was appropriate and beneficial. The methodology employed was basically skimming and manual removal of stranded oil and contaminated sand from the sandy beaches. The main thing that would have improved the cleanup, in my opinion, would have been selling the oiled sand to an asphalt company for road construction rather than hauling it to a Class 1 landfill [12] and using up valuable landfill capacity.

Table 1

Distribution Of Sediment PAH By Treatment Category And Elevation In Prince William Sound In 1990

Average PAH Concentration ± SD (ppm dry weight of sediment from July And September [4]

Elevation	SITE OILING CONDITION Control Sites	Oiled & Untreated Sites	Oiled & Treated Sites
Upper	0.0018± 0.0033	3.4± 6.4	0.65± 1.03
Mid	0.00077± 0.00062	16.2± 45.0	1.8± 4.1
Lower	0.0024± 0.0025	0.79± 1.83	5.8± 17.7
Shallow Subtidal	0.012± 0.006	0.45± 0.11	3.5± 3.0
Overall	0.0022± 0.0036	6.5± 26.1	3.2± 11.0

REGULATORY LESSONS

Several regulatory lessons can be learned from the events surrounding the EXXON VALDEZ oil spill.

Poor Implementation of Existing Pre-spill Planning and Requirements. Because traffic control radar systems were below specifications, the Coast Guard was unable to detect departure of the EXXON VALDEZ from the traffic lanes and thus failed to provide a warning to the ship. The system had been downgraded to save money, despite near major accidents on at least two previous instances since 1976 (e.g., the SUN PRINCE WILLIAM SOUND in 1980). Requirements for tug and pilot accompaniment to the Hinchinbrook Entrance to the sound had also been downgraded because of cost. Moreover, state and federal government allowed the industry to continue operations in the absence of adequate quantities of operational cleanup equipment, despite repeated warnings from one state inspector in Valdez.

Based on these decisions or lapses in enforcement, it could be argued that federal and state governments should share the blame and liability with the offending industry; after all, government's mandate is to protect the public trust rather than facilitate exploitation of one resource at the expense of another.

Scarcity of Experienced or Appropriately Trained Staff for Specifics of the Problem. Trained, experienced personnel of all types were very scarce and were generally involved elsewhere. NOAA, represented by the Hazardous Materials Response Branch, probably had the largest cadre of trained people. The other state and federal agencies were generally poorly staffed for response to such an emergency, and several federal agencies that had people with substantial background in oil spills declined to become involved in the spill. It is likely that less than five percent of the management personnel sent to respond to the EXXON VALDEZ oil spill had prior experience with an actual oil spill or training appropriate to their particular responsibilities in the response. Thus, most were unprepared, poorly trained, and were developing the personal philosophy with which they were going to approach this crisis of epic proportions while on the job. In general, most had a strong belief that oil was bad and, on that basis, believed that all oil had to be removed. Overall, there was a serious lack of objectivity. Few questioned the necessity of the cleanup, evaluated objectively the condition of the biota in oiled areas, or calculated what damages the cleanup itself might wreak. These views are in keeping with the observations of Lindstedt-Siva that the tendency during implementation of a cleanup is returning to a goal of removing all oil from the environment rather than heeding the sensitivities of the biological, socioeconomic, and aesthetic needs of the damaged system [11].

Criteria for Need and Completeness of Treatment are Inappropriate. In early spills, criteria for effectiveness and completeness of a cleanup were based on the concentration or appearance of oil. During a later period, federal agencies were given the responsibility for advice and oversight, whereafter biological criteria became more important. In the most recent spills, however, state agencies and public interest groups have demanded a role, resulting in greater influence on the process from people with less experience and objectivity and less exposure to appropriate literature or the manner in which the environment responds to oil. As a consequence, decisions have again become based more on the amount of oil remaining in habitats than on the nature, condition, and amount of biota residing in the contaminated areas or other long-term considerations [11].

The need to integrate trained biologists into response planning is fundamental if a major objective of treatment efforts is to protect the biota. This has to be more than a token involvement; trained, experienced biologists should have at least an equal vote. In most areas, it is mainly the biota that we are trying to save and protect and biologists have to be an integral part of the planning and assessment effort. Biologists are better trained than engineers, chemists, geologists, or military personnel for developing treatment programs that are responsive to the needs, tolerances, and sensitivities of the biota. It is likely that many inexperienced biologists also lack this insight. In my opinion, actions in Prince William Sound provide abundant evidence of this.

The primary goal of a cleanup should be to protect economic, biological, or aesthetic resources. The benefits of removing oil from the environment must be weighed against both the potential environmental and economic costs of removal. Removal purely for emotional or short-term aesthetic reasons is generally not an adequate justification for the biological or economic costs. The revised objectives should be used as the basis for establishing relevant criteria by which a cleanup is planned and its effectiveness and completeness are measured.

Absence of Independent Oversight of Cleanup and Evaluations of Proposed Cleanup Methodologies. A requirement for environmental impact analyses should be established and implemented for all potential methods of oil spill treatment, especially in critical habitats. Where possible, these analyses should be completed on a regional basis prior to any spill events. In the crisis mode that exists at most spills, the cleanup operators and managers are grasping at straws in the face of overwhelming odds and are not willing to conduct environmental impact assessments for potential methodologies. The result can be imposition of substantial additional insult on an already injured biota through implementation of harmful treatment schemes.

Following development of information on the impacts of various treatment methodologies, agencies and spill monitors need to develop a good understanding of treatment effects for the various potential methodologies so that cleanup strategies can be developed that consider the benefits, costs, advantages, and disadvantages of proposed alternatives on the various components of the biota.

The much vaunted bioremediation methodology developed and implemented by the EPA and Exxon during the EXXON VALDEZ cleanup was not subjected to careful impact assessment to determine its effects on the benthic biota in the areas to which it was applied directly. Moreover, toxicity testing conducted on seawater collected adjacent to a treated beach indicated that mortality and frequency of abnormal development in larval oysters became increasingly higher for at least 18 hours following application during low tide (Figure 5; [13, 14]). Despite the fact that data available at the time did not show that the methodology was any more effective at oil removal than the naturally occurring bacterial populations that developed in response to the oil [14], the decision was made in August to bioremediate over 70 miles of shoreline. Thus, the appointed regulators (in this case, EPA) were also part of the development team, had a severe conflict of interest, and failed in their mandate.

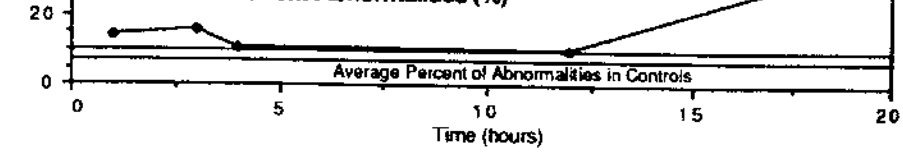

Figure 5. Survival and frequency of abnormalities in oyster larvae exposed to Inipol/seawater solution of an Inipol-treated beach in Prince William Sound (based on Pritchard et al. 1990).

The lack of an objective viewpoint during the EPA study is demonstrated by the manner in which its objectives were defined. As stated by Sanders and Gray, "The objectives of the project are to demonstrate the effectiveness of adding nutrients to enhance natural microbial oil degradation as a method of contaminated shoreline cleanup and to evaluate environmental side effects of the process" [14]. This wording of the objectives tends to suggest that the conclusions of the studies were pre-determined.

Our findings suggest that the intertidal biota was damaged because of the lack of an effective mechanism in the bureaucracy to ensure that independent and enlightened damage assessment is fed back to the spill managers (the FOSC). While numerous well trained people observed and reported considerable damage resulting from the cleanup activities, there were no lines of communications to the people "driving the tank" and no strong reason for the "drivers" to pay attention. As EPA's role in the development of the bioremediation program shows, the system needs an independent environmental watchdog, possibly a committee from the National Academy of Sciences or some other distinguished organization with considerable influence, a great deal of academic prestige and objectivity, and a minimum of political baggage. Moreover, this group should be required to evaluate all of the proposed cleanup methodologies prior to approval of wholesale application. This should exclude state and federal agencies such as EPA because most of the senior managers in these agencies are highly political and may be more interested in improving the influence of their agency than in the wisdom with which a crisis operation is conducted.

TREATMENT LESSONS

At least three types of lessons can be learned from analysis of the treatment operations for the EXXON VALDEZ spill. First, the adequacy and effectiveness of containment and capture equipment is generally poor relative to the magnitude of a major spill. As a colleague commented while flying over the sound in the early stages of the spill response, the effectiveness of a skimmer operating in the middle of a major slick appeared similar to trying to trim a golf course with fingernail clippers. Data from past spills suggest that the effectiveness of various containment and capture methodologies is low. Generally, the best that can be achieved is 10 to 15 percent recovery. The remaining 85 to 90 percent of the oil remains in the environment, either as gases (fumes) in the atmosphere, deposits on the beach or the sea floor, solutes in the water, or solids (tarballs) at the surface of the open ocean; up to 60 percent may be unaccounted for [5, 15].

A second lesson is that shoreline treatment can cause a substantial amount of additional physical and ecological damage to the affected habitats. Recent and past studies provide abundant evidence of this and contribute ample cause for review of all post-spill planning documents to assess the costs and benefits that might be derived from treatment operations [2, 3, 4, 5, and 14].

A third lesson that can be gleaned from both the EXXON VALDEZ and AMERICAN TRADER spills is that physical removal of oil or asphalt can be beneficial from either a biological, economic, or aesthetic viewpoint. This is particularly true of substrates that are characterized by ephemeral biological assemblages, for example, sand or gravel beaches. However, cleanup procedures should be carefully evaluated where long-lived mature of plants or animals are significant components of the affected communities and would be removed or damaged by the removal of oiled sediments [4, 15].

SCIENTIFIC LESSONS AND PROBLEMS

The types of scientific lessons that we learned from participation in the scientific effort on these spills emanated from problems associated with development and implementation of suitable sampling designs. Specifically, problems that confronted us included: 1) availability of adequate or timely baseline data, 2) availability of undisturbed oiled sites at which to monitor damage from the contamination and recovery of the systems, as well as comparative areas for evaluating the effects of oiling and treatment, 3) availability of public funds to support independent studies of impacts and recovery, and 4) the effects of the NRDA process on an ecological assessment of effects, processes, and recovery.

Absence of Baseline Data in Areas Where Spills are Likely. Establishing a suitable sampling design is usually hampered by several types of difficulties. One of the more burdensome is the general absence of current (or even dated) baseline data, both regionally and at specific sites, for the abundance, distribution, and condition of communities and populations that may be affected by the spill. Spills often occur in regions where baseline data are lacking and response occurs too slowly to permit acquisition of good baseline data. This presents major problems for sampling design for either effects or NRDA studies.

Sampling Design Problems. The fact that the spill is not replicated creates a problem for people ascribing rigorously to the pitfalls of pseudoreplication [16]. It also causes substantial difficulties for people trying to operate with the BACI design described by [17]. Achieving an adequate design that deals with all of the uncontrolled variables that occur in nature is a considerable challenge on short notice in any event. Another challenge is the uncertainty associated with selection of sites prior to their being oiled in order to provide a suitable representation of oiled and unoiled sites and, if treatment becomes an issue, of oiled and treated sites.

Since Hurlbert pointed out the problems of pseudoreplication in sampling design [16], many of the traditional approaches to assessing impacts have become questionable and a variety of new methods and approaches to sampling design and data analysis have been developed and must be considered in order to enhance the scientific acceptability of the sampling program that is ultimately adopted [e.g., 18]. Pseudoreplication is an issue that must be dealt with in site selection and it raises some knotty complications in developing a strategy for stratified randomization, replication, and allocation of resources (i.e., the trade-offs between numbers of sites, numbers of replicates per site, and budgets available for time and funding).

Unwillingness Among Regulators and Cleanup Operators to Concede Adequate Site Set-Asides and Funding for Scientific Evaluation of Oil Spill Damage and Treatment Effects. In the case of the EXXON VALDEZ spill, state and federal regulators displayed a strong reluctance to allow oiled areas to remain undisturbed as treatment set-asides in which research could be conducted to distinguish between the effects of oiling and treatment in various habitats. For Prince William Sound, a maximum of twelve small sites (each about 75 m of shoreline) was ultimately left untreated to permit scientific studies; only a few of those were suitable for biological studies. This also creates difficulties in sampling design because of the limitations in suitable replication for untreated habitats in each substrate category. The unfortunate consequence of this shortsightedness is that we as a society will come away from this event having learned only a fraction of what we could have gleaned to improve our response to future spills and it is clear there will be further spills in the future [7].

The paucity of public funding for independent research has also been a matter of some discussion [e.g., 19]. It appears that our NOAA investigation and a study by Juday of the University of Alaska are the only on-going publicly funded programs for the EXXON VALDEZ oil spill. It appears that no studies have been publicly funded to assess long-term impacts of the AMERICAN TRADER spill. The National Science Foundation does not fund such research, particularly in a manner responsive to an emergency situation. A group representing the National Academy of Sciences visited Prince William Sound early in the event but no studies materialized. The federal and state agencies that conduct the studies to assess the efficacy of these projects in the first place (e.g., MMS) do not fund studies to assess the impacts of accidents resulting from those projects. I have been told by several agency personnel that no agency has responsibility for evaluation of effects from oil spills in open waters. The great majority of funding for the EXXON VALDEZ studies, whether conducted by the state or federal agencies, universities, or consultants, ultimately came from Exxon. Most of this research was aimed at supporting a determination of the value of the damages as part of the NRDA process and it is likely that most of these data will never be evaluated and released in peer-reviewed journals. Moreover, the studies were focused on assessing the dollar value of damage rather than on examining alterations in biochemical, physiological, and ecological processes. Consequently, a great deal concerning the causes of damage and the chronic effects of the spill and the ensuing treatment was never addressed. The valuable innate curiosity of the scientist was blunted or diverted.

NRDA — Constraints to Scientific Exchange of Ideas. A great amount of funding was directed toward assessing for the Potentially Responsible Party (Exxon) and the Trustees (federal and state agencies) the damages to the natural resources. This process, termed the Natural Resource Damage Assessment (NRDA), is focused on acquiring data for use in court proceedings and litigation; access to and discussion of the information obtained during this process is very difficult. We found that the scientific process and transmittal and exchange of scientific observations and ideas did not work well in an environment riddled with attorneys and the NRDA process. Articles discussing the effects of the NRDA process on the scientific process and studies of the effects of the spill and ensuing recovery of the biota have just started to appear in the media [e.g., 20, 21] but it is clear that movement to separate the scientific and NRDA process is gaining momentum [11].

DATA GAPS

Based upon our experience with the EXXON VALDEZ and AMERICAN TRADER spills, I have a sense that several types of information necessary to conduct accurate assessments of the effects of an oil spill are lacking in most areas with a higher-than-average likelihood of experiencing an oil spill within the foreseeable future.

Baseline Knowledge. I have mentioned above the inadequacy of baseline data in many places where oil spills are a reasonable likelihood. The common argument given in opposition to conducting baseline studies in such areas is high cost. Nevertheless, both government and industry make a great deal of money off the lease of oil-producing properties and the sale of the resources themselves. Considering the costs of the EXXON VALDEZ cleanup and the minimal effect these expenditures had on Exxon's profits in 1989 and 1990 (see discussion below), the costs of supporting monitoring studies in these areas are small indeed compared to the profits yielded and the value of the baseline information when an accident occurs. The cost argument appears specious at best, especially considering that monitoring costs generally would be split among several operators. Tax or rate payers routinely pay for monitoring programs for most ocean outfalls and it does not appear unreasonable to require ongoing baseline studies in areas where the risk of large accidental release of oil is high.

Definition of "How Clean is Clean?" A question that is heard frequently around an oil spill is, "How clean is clean?" The real meaning of the question is, "How much treatment is enough?" Treatment and cleanup methodologies comply with the law of diminishing returns; effectiveness declines with increasing duration of activity. However, daily costs are probably fairly stable. Thus, one needs to ask, "When does the treatment cease to be effective or begin to create more problems than the spill?" Also, at what point does the cost of the treatment exceed the ecological benefit of the treatment? However, these questions need to be answered before the spill occurs so the answers can be agreed upon before the situation becomes superheated and the participants become strongly polarized by political, emotional, and economic pressures.

Effects and Effectiveness of Chemical Dispersants and Bioremediation Agents. Well designed, comprehensive studies need to be conducted and reviewed by independent organizations to determine the effectiveness and ecological effects of dispersants and bioremediation agents. A large proportion of the available studies have been prepared by groups with a distinct conflict of interest and need to be re-examined. Moreover, the information available is dominated by laboratory studies and few data are available on either acute or chronic community effects. The need for such independent research is driven home by the manner in which the EPA handled the bioremediation research for the EXXON VALDEZ oil spill.

FURTHER DISCUSSION

The analysis above suggests a need to ask some important questions before developing oil spill policies and contingency plans. Important questions specific to areas of elevated risk of spills include: What resources are at risk and why are we concerned for them? Do we know enough about the manner in which these resources respond to oil spills and cleanup technologies? What are the real (rather than purported) economic and environmental costs of a spill and the costs and benefits of a cleanup? Will reparations for damages be sought through litigation? How does the cost of litigation compare with the cost of the lost resources? Are we sufficiently concerned about the environmental costs of a spill to commit to the funding necessary to obtain good information on spill and treatment effects necessary to support successful litigation? Who pays the bills for the cleanup, restoration efforts, and the litigation? What is the cost-benefit ratio?

Arguments that adequate marine safety programs, double-hulled tankers, baseline studies, treatment effects studies, etc. are too expensive are generally inaccurate and self-serving. The cost of the EXXON VALDEZ cleanup far exceeded the potential costs for: 1) restoring or upgrading the safety program called for in the original agreements and contingency plans for Port Valdez, 2) conducting adequate baseline studies, or 3) evaluating the effects of various treatment options. Nevertheless, according to the Value Line and the Standard & Poor stock reports, Exxon's profits for 1989 ($4.55 billion) were only slightly (11 percent) lower than profits for 1988. Despite that decline, dividends were increased by 7 percent [22, 23].

CONCLUSIONS

Following is a brief summary of some of the lessons offered by the EXXON VALDEZ oil spill experience:

Generally,

The acute effects of an oil spill may be severe but are of a relatively short duration;

- The major effort should be expanded in collecting oil from the water surface while it is concentrated and prior to shoreline impact;
- Shoreline cleanup efforts following a spill can be helpful under certain circumstances but often cause greater and longer term damage than the spill itself;
- The objectives of a cleanup should be determined and clearly defined in contingency plans for each area at risk before there is a problem; cleanup methodologies compatible with these objectives should be identified and described therein;

- Cleanup efforts should be initiated on the basis of a balance of biological, economic, and aesthetic criteria rather than in response to raw chemical or visual criteria, or public and political outcry;

- Criteria by which the effectiveness of a cleanup is measured should reflect the value of the biological, economic, and aesthetic resources of the spill area;

- Monetary and biological costs of a cleanup should be weighed against the potential benefit to the biota or human activities; and

- An unbiased, knowledgeable, and independent environmental watchdog committee should review and evaluate all proposals and supporting data for treatment alternatives before implementation is permitted on a large scale.

For the EXXON VALDEZ oil spill,

- Although some types of treatment were justified in some locations, the bureaucracy ignored the abundant lessons of oil spill history;

- A significant amount of money was spent to no useful end or to further damage already damaged habitats;

- Treatment was probably innocuous but unnecessary over a significant proportion of the shoreline;

- Treatment was highly detrimental over a significant proportion of the shoreline; and

- Many state and federal agencies were a major part of the problem rather than the solution during the cleanup.

ACKNOWLEDGEMENTS

I wish to acknowledge Dr. Jonathan Houghton, Mr. William Driskell, and Mr. David Erikson for discussions and moral support while we passed through a biologist's version of Dante's Inferno engendered by the experience of and activities surrounding the EXXON VALDEZ oil spill. I and my colleagues are grateful for the funding from 1) Exxon for our field studies in 1989, and 2) NOAA, MMS, EPA, USCG, API, and MSRC in 1990 and 1991. Finally, I wish to thank Dr. June Lindstedt-Siva, Lt. Cdr. Gary Petrae, Mr. Gerald Schimke, Mr. Jack White, and Dr. Houghton for critical review of the manuscript, and Dr. Alan Mearns, Mr. Gary Shigenaka, and Ms. Rebecca Hoff of the NOAA Biological Assessment Team for discussions and provision of materials regarding bioremediation

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