JOY B. ZEDLER, PRINCIPAL AUTHOR

Tidal Wetland RESTORATION

A Scientific Perspective and Southern California Focus

A PUBLICATION OF THE CALIFORNIA SEA GRANT COLLEGE SYSTEM

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Published by the California Sea Grant College System, University of California, La Jolla, California, 1996. Additional single copies are available for $10 from: California Sea Grant College System, University of California, 9500 Gilman Drive, La Jolla, CA 92093-0232. (619) 534-4444.

This publication is funded by a grant from the National Sea Grant College Program, National Oceanic and Atmospheric Administration, U.S. Department of Commerce, under grant number NA36RG0537, Project number A/P-1. The views expressed herein are those of the author and do not necessarily reflect the views of NOAA or any of its sub-agencies. The U.S. Government is authorized to reproduce and distribute for governmental purposes.

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A Scientific Perspective and Southern California Focus

Joy B. Zedler, Principal Author

Pacific Estuarine Research Laboratory
San Diego State University
San Diego, California

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University of California
La Jolla, California 92093-0232
Publication No. T-038

ISBN No. 1-888691-02-6

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*Cite as: Zedler, Joy B., Principal Author. 1996. Tidal Wetland Restoration: A Scientific Perspective and Southern California Focus. Published by the California Sea Grant College System, University of California, La Jolla, California. Report No. T-038.

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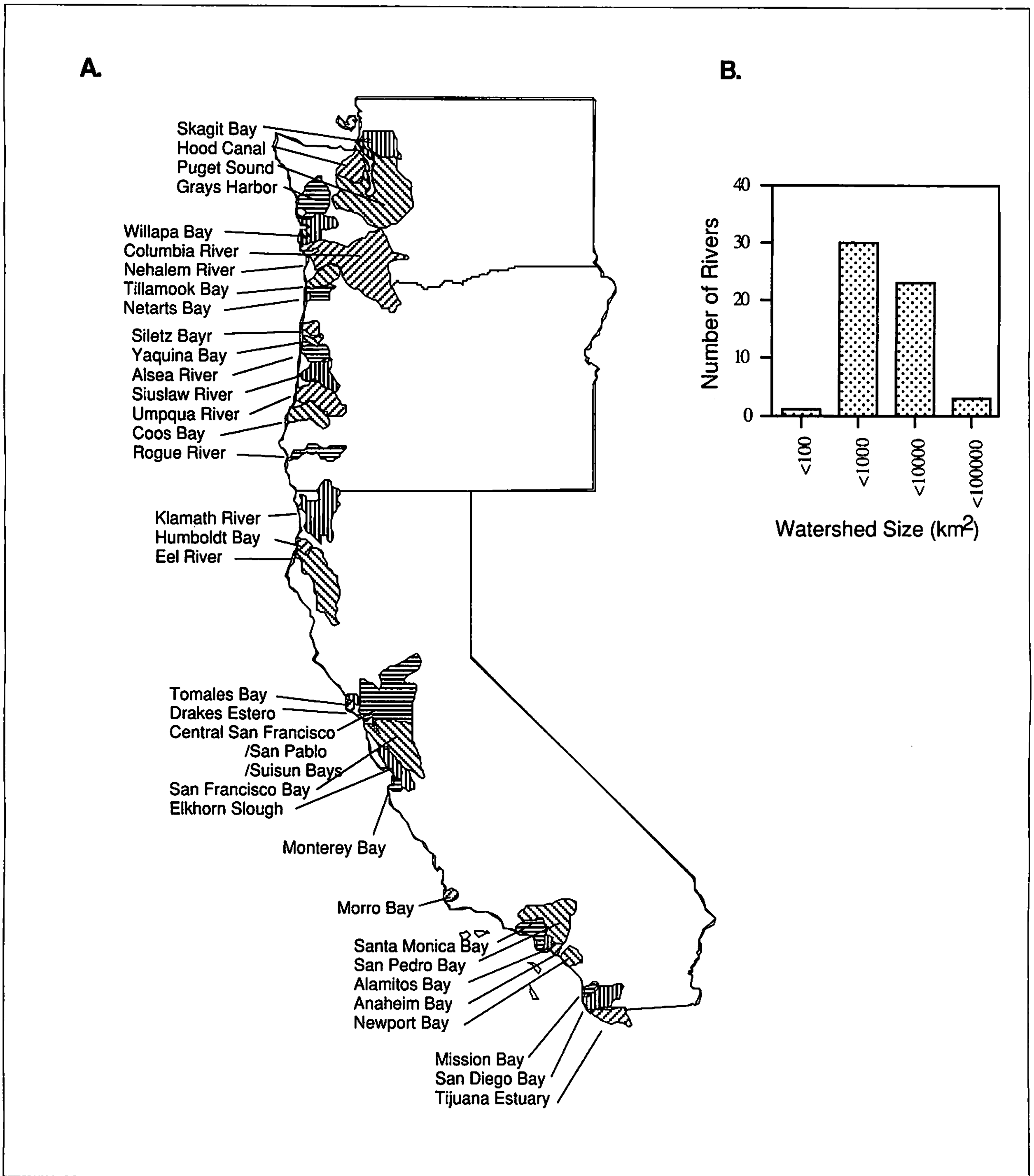


Figure 1.1 West Coast watersheds. A. Estuarine drainage areas of the Pacific coast of the conterminous United States. Areas are lower parts of the watershed that drain directly into estuarine waters and have the greatest human influence. From NOAA 1990. B. Sizes of coastal watersheds in California. Data from Jacobs et al. 1993.

PREFACE

This book is an effort to synthesize knowledge and make scientifically based recommendations for improved wetland management. It provides recommendations for management based on what I and my associates at the Pacific Estuarine Research Laboratory (PERL) have learned about the functioning of wetland ecosystems. Studies by many graduate students and research associates at PERL and funding from both public agencies and nonprofit foundations contributed to the information presented here.

My understanding of regulatory issues comes from interaction with many excellent employees of the California Coastal Commission, the State Coastal Conservancy, the U.S. Fish and Wildlife Service, the U.S. Army Corps of Engineers, and the Environmental Protection Agency. Many of my students have gone on to governmental positions or become environmental consultants; from them I have learned how important decisions are often made with insufficient scientific understanding. I have also benefited from the views of public volunteers. Southern California is probably unique in having such a large number of citizen groups that focus on their respective nearby wetland, for example, Friends of Famosa Slough and the Southwest Wetlands Interpretive Association. I have learned a great deal from these dedicated citizens about what people want from wetlands. My interactions with scientists around the world have provided many opportunities to discuss how to manage urban and agricultural development in ways that help sustain, rather than jeopardize, regional biodiversity. My understanding of agriculture and attitudes about property rights is rooted in the midwestern farming community where I was raised. As a result of all these influences and interactions, I believe I speak as a practical scientist with a love of the land and a strong desire to sustain the earth's biodiversity.

This preface provides an opportunity to explain how the wetland research program came to be and to offer thanks to everyone who played a role in its development. PERL was established in 1984, when funding was made available by the California State Resources Agency to create freshwater wetlands for habitat and outdoor mesocosms for research on abandoned agricultural land adjacent to Tijuana Estuary. A field research laboratory was developed on a 28-ha site purchased by the State Coastal Conservancy for the Tijuana River National Estuarine Research Reserve, with title given to the City of San Diego. The U.S. Fish and Wildlife Service contributed two surplus trailer houses that became the first "labs." We installed some fencing, laid a water line, ran electrical wiring to the site, and had signs printed; San Diego State University recognized the value of the activities by making the laboratory an official unit of the Department of Biology and College of Sciences. Interaction and communication between agencies and researchers have been close throughout the history of PERL, with at least six agencies involved in setting up the initial facility. As research activities expanded, the name PERL also expanded to mean the research group that actively studies wetlands. Just 8 years later, the San Diego City Council declared PERL Day on July 6, 1992, in recognition of services provided to the public, especially information relevant to the management of Tijuana Estuary.

Having directed the group since its inception, I am in debt to hundreds of people who believed in the importance of science-based decision making; who asked for advice; who wrote letters of support for our research proposals; who encouraged our resubmissions when original submissions were rejected; who held a shovel, refractometer, or clipboard; who manned fish seines and counted plants; who probed mud samples for microscopic beasts; who spent long hours at the computer making sense of mud-stained data sheets; who helped write innumerable reports; and who helped interpret the wonderful complexity of Southern California's coastal wetlands. In deep gratitude, I dedicate this book to all for their help, with special thanks to those who have continued in careers using wetland science to improve the management of natural resources.

The mainstay of PERL's research funding base has been the California Sea Grant College System. When Tijuana Estuary was designated a national reserve, the Sanctuaries and Programs Division of the National Oceanic and Atmospheric Administration (NOAA) added research and monitoring funds for work at that site. The U.S. Fish and Wildlife Service (now the National Biological Service) established a cooperative agreement that provided equipment and researchers to assist with restoration research. NOAA's Coastal Ocean Program supported complementary studies of ways to accelerate the development of constructed wetlands. The California Department of Transportation (Caltrans) supported monitoring work at San Diego Bay and facilitated research at the department's mitigation site. Los Peñasquitos Lagoon Foundation and San Diego County funded monitoring of two coastal lagoons north of San Diego, both for the purpose of understanding problems caused by closure of ocean inlets. Many of the research grants have required nonfederal matching funds, which have been generously provided by the College of Sciences and Department of Biology at San Diego State University.

Reviewers contributed significantly to this effort. The contributors and I are grateful to Si Simenstad, Wayne Ferren, and Joanne Kerbavaz for their time and thoughtful criticisms. We are grateful to the California Sea Grant College System for support of this publication, and especially to Rosemary Amidei, communications coordinator; Barbara Halliburton, copy editor; and Joann Furse, for desktop publishing.

Joy Zedler, PERL Director

CHAPTER 1

Pacific Coast Estuaries and Wetlands

1.1 INTRODUCTION

The U.S. Pacific Coast parallels a major mountain chain and a steep continental shelf. Watersheds are small and close to the coast, with two conspicuous exceptions: those of the Columbia River and San Francisco Bay (Figs. 1.1A and B). Coastal streams are numerous (more than 100 in California alone) but rarely extensive. Rainfall is highly variable, both temporally and spatially, in many coastal areas (Fig. 1.2). Most of California is characterized by a "Mediterranean-type climate," with warm, dry summers and cool, moist winters. Summer always includes a period of little or no rainfall (Fig. 1.3). Even the Pacific Northwest experiences summer drought. The noncontiguous states have climatic extremes, with shortened growing seasons and colder temperatures in Alaska and tropical heat and high year-round rainfall in Hawaii. The diversity of estuarine types within the Pacific states is high, because of both physiographic and climatic variability—for example, fjords in Alaska; mangrove-dominated sites in Hawaii; major bays in Puget Sound, Humboldt Bay, San Francisco Bay, San Pedro Bay, and San Diego Bay; and intermittent lagoons between these landmark bays. Despite their northern latitude, few of the Alaskan estuaries freeze over in winter, so ice damage is uncommon.

1.1.1 Comparisons with Other Coasts

On the Atlantic side of North America, the United States spans a narrower latitudinal range than on the Pacific, and the coastal topography is much more gentle. Estuaries in the East have been studied much longer than those in the West, because estuarine science developed on the Atlantic Coast. Until recently, an entire scientific journal, *Chesapeake Science*, was devoted to studies of the largest and most valued U.S. estuary, Chesapeake Bay. The broad Atlantic coastal plain has many large watersheds, and most of the rivers have continual inflows of fresh water, because summer rainfall is ample. Together, the large watersheds and year-round precipitation produce the "textbook estuary": one with a gradient of salinities from fresh to marine, a rich complement of organisms that are adapted to brackish water, and highly productive tidal marshes that export substantial quantities of particulate organic matter to nearshore waters and provide the food base for large numbers of commercial fish species.

The U.S. estuaries house large areas of intertidal wetland, but the East and West coasts differ in habitat distribution. According to Field et al. (1991), the conterminous United States has more than 1.6 million ha of coastal salt marsh, of which only 3% occurs along the Pacific Coast. Tidal flats cover a smaller area nationwide (approximately 0.5 million ha), but the Pacific Coast accounts for 17% of that total (Table 1.1). Thus, Pacific intertidal habitats are less vegetated than those along either the Atlantic or the Gulf coasts. The Pacific Coast has 1.64 times as much tidal flat as salt marsh, compared with a ratio of 0.24 for the rest of the conterminous United States. No doubt this reflects the geologically younger, more tectonically active, and steeper Pacific coastline.

1.1.2 Tidal Patterns

Twice a day, ocean waters enter and leave the tidal wetlands along the Pacific Coast. This pattern is a semidiurnal tide, in which the rising or flood tide is followed about 6.2 hr later by a falling or ebb tide. California's coast experiences "mixed semidiurnal tides," that is, a complex tidal pattern with two high tides per day that are unequal in height and two low tides per day that are also unequal (Fig. 1.4). Each day, the higher high tide occurs 50 min later in the day than the day before. Spring tides (those with greater amplitude) occur when the forces of the sun and moon act together and alternate with neap tides, which occur every other week when the gravitational forces of the sun and moon oppose each other. (For a complete discussion of tidal forces, see Defant 1958.)

In Southern California, the highest tides (extreme higher high water) are about 7.5 ft (2.3 m) above the reference datum (mean lower low water = 0); mean tidal amplitude is about 3.9 ft (1.2 m), and mean spring tidal amplitude is about 5.6 ft (1.7 m) (NOAA 1995). It is useful to summarize tidal data on a monthly basis to show the extreme high water, the mean higher high water, and the number of times that tides exceed various elevations (Table 1.2).

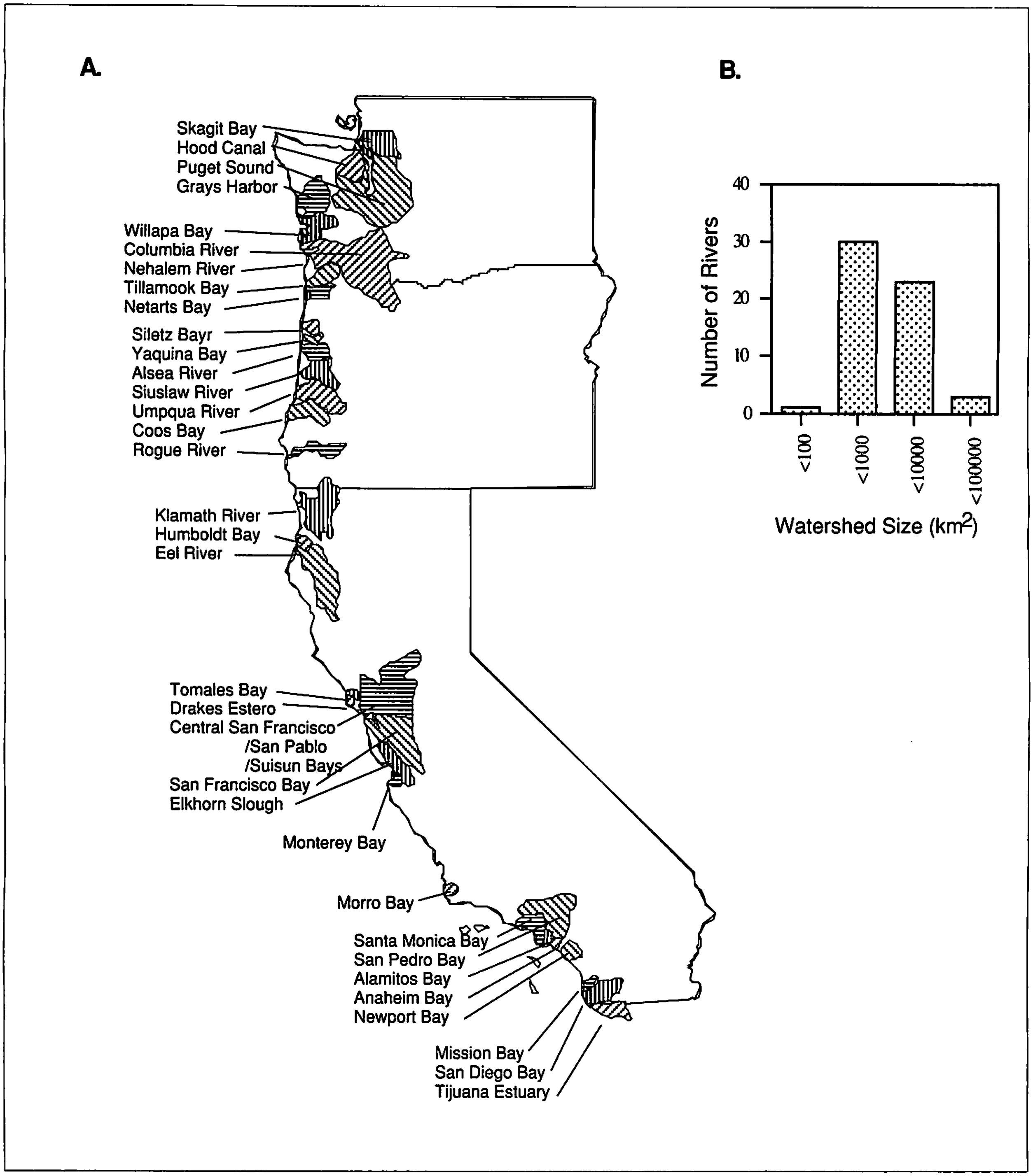


Figure 1.1 West Coast watersheds. A. Estuarine drainage areas of the Pacific coast of the conterminous United States. Areas are lower parts of the watershed that drain directly into estuarine waters and have the greatest human influence. From NOAA 1990. B. Sizes of coastal watersheds in California. Data from Jacobs et al. 1993.

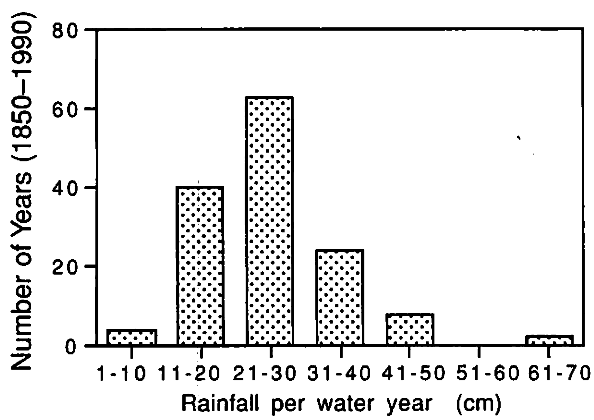


Figure 1.2 Histogram of rainfall at San Diego (Lindbergh Field) over a 140-year period. Water years are October 1 through September 30.

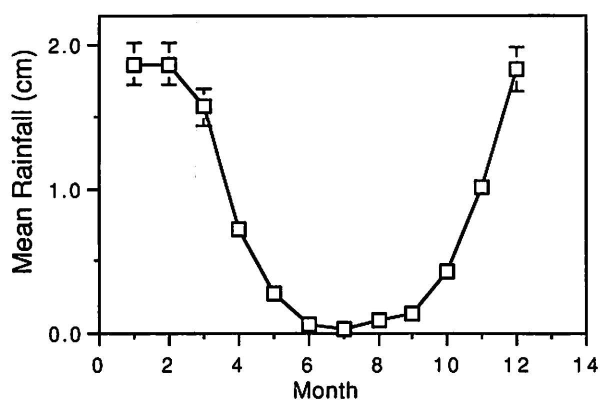


Figure 1.3 Mean rainfall per month for San Diego (Lindbergh Field) for 1850–1990. Bars = SE.

In December, the mean higher high water is maximal; in March, it is minimal. Few high tides occur in March. Several aspects of this tidal pattern have special significance for wetland biota. Salt marsh algae and vascular plants are highly dependent on tidal inundation. It is clear from their phenology that the vascular plants depend on tidal water for moisture. They grow actively through summer when Southern California has no rainfall. In immediate contrast, the plants of the adjacent upland are active in winter and dormant in summer, as is the coastal sage scrub and chaparral of this region. At the wetland-upland boundary, *Salicornia subterminalis* (a wetland obligate) is active in the summer, whereas *Lycium californicum* (an upland shrub) is dormant. These two plants often grow side by side, but they obviously differ in their water sources, with *S. subterminalis* able to tap into saline groundwater during the long, dry summers.

Marsh vegetation grows most actively between March and June (Winfield 1980), perhaps in response to increasing length of day and warmer temperatures, but the daytime low tides in spring (Fig. 1.4) should act synergistically to warm soils and stimulate growth of roots and rhizomes. The light-footed clapper rail *(Rallus longirostris levipes)* appears to take advantage of this pattern by building its nests in the lower marsh beginning in February or March. Nests are least likely to be inundated at this time (Zedler 1993). Macroalgae grow rapidly in the spring, when daytime low tides provide more light (Rudnicki 1986).

In the fall, low tides occur during the late afternoon, whereas in the spring, the lowest tides occur at night. Low tides during hot afternoons lead to warm and hypersaline marsh soils. When the next tide covers the marsh, the water absorbs the heat, and channel waters and associated fauna are warmed by the ebb tide. Water temperatures are greatest at this time, in part because of the long exposure to sunlight but also because of water contact with warm marsh soils. This may be one reason use of the region's intertidal marshes by fish is limited. PERL has begun continuous monitoring of water salinity, temperature, pH, and oxygen at Tijuana Estuary, so that these long-recognized patterns can be documented.

Because tides have such an enormous influence on coastal wetland biota, the need is great to understand further the dependencies on and tolerances of the timing and duration of tidal inundation and exposure (hydroperiod) of various species. Because of the seasonal and daily variations in hydroperiods, this is not a simple task. Most of our understanding is based on comparisons between sites with a long history of good tidal flushing and sites with interrupted tidal action. Tijuana Estuary (section 1.1.3) is a good example of the former type; it supports more native species than less tidally influenced systems (section 2.1).

1.1.3 A Sample Pacific Coast Wetland

In San Diego County, the estuarine "type specimen" is Tijuana Estuary, and conditions at this tidal system illustrate the environmental forcing functions that control the biota. Tijuana Estuary (lat 32°34'N, long 117°7'W) is the southwesternmost estuary of the continental United States. Because of its location on the tectonically active Pacific Coast, it has a dynamic geologic setting. This drowned-river-type estuary is small (1024 ha) compared with river mouths along the more gentle topography of the Atlantic and Gulf of Mexico coasts. Its river floods only during years when large winter storms occur; even then, it is virtually dry during summer.

The 448,323-ha watershed is mostly (73%) within Mexico. Together, one large reservoir, behind Rodriguez Dam on Tijuana River, and two small U.S. reservoirs on Cottonwood Creek, control 78% of the watershed upstream of Tijuana Estuary. Situated 8 km upstream of the estuary is the metropolis of Tijuana, Baja

Table 1.1. Comparison of Salt Marsh and Tidal Flat Areas in the United States, by Region

Region	Area: Salt Marsh		Area: Tidal Flats		Ratio of Tidal Flats to Salt Marsh
	Acres	Ha	Acres	Ha	
Atlantic	1,651,900	668,515	500,500	202,550	0.30
North Atlantic	(68,100)		(142,800)		
Middle Atlantic	(689,600)		(256,100)		
South Atlantic	(894,200)		(101,600)		0.20
Gulf of Mexico	2,496,600	1,010,360	507,500	205,382	0.20
Pacific	121,900	49,332	200,100	80,979	1.64
Washington	(18,300)		(75,200)		
Oregon	(13,200)		(30,900)		
California	(90,900)		(93,900)		
Total for Nation	4,270,300	1,728,167	1,208,000	488,871	

Note: Numbers in parentheses are subtotals for subregions. Acreage data from Field et al., 1991.

Table 1.2. Number of 1990 Tides Predicted for San Diego Bay (Broadway Pier) That Would Exceed Selected Elevations

Month	Number of Tides Per Month That Exceed Each Datum: >210 (>6.9)	>195 (>6.4)	>180 (>5.9)	>165 (>5.4)	>150 (>4.9)	MHHW (cm)	EHW (cm)
Jan	5	12	15	19	26	179	226
Feb	1	3	15	24	30	174	210
Mar	0	3	9	25	32	163	201
Apr	3	5	7	17	28	168	216
May	5	6	13	18	23	172	229
Jun	6	7	15	21	25	184	232
Jul	5	12	17	22	33	184	232
Aug	3	11	16	31	36	181	219
Sep	0	5	21	32	39	180	198
Oct	3	8	17	30	39	179	213
Nov	5	12	18	24	25	184	229
Dec	7	13	20	24	28	188	238

Note: MHHW = mean higher high water; EHW = extreme high water. Data summarized from National Ocean Service tide tables (from Zedler 1993).

California, a growing city of about 2 million inhabitants. Between Tijuana and the estuary is an urbanized and agricultural river floodplain with no pristine habitat.

Southern California has a dry Mediterranean-type climate, with mild, wet winters and warm, dry summers. Annual rainfall is usually low, with an average of approximately 25 cm on the coast (Fig. 1.2) and 50–100 cm in the surrounding source mountains (Pryde 1976). The average monthly temperature range is only from 9–21°C (48°–69°F, NOAA data for San Diego), and average daily solar insolation is 178–606 langleys (Taylor 1978). Frost is extremely rare along the coast, and snow is virtually unknown. However, although temperature is predictably moderate, rainfall and streamflow are not. Neither the amount of freshwater inflow nor its timing is dependable.

Precipitation data suggest that rainfall increases gradually from October to a peak in January and February and then gradually declines through April. Averages are misleading, however, because a single month may include much of the year's total rainfall. In December 1921, rainfall was 24 cm, nearly equal to the annual average. The month with the most variable rainfall is December (standard error [SE] = 0.4); the least variable month is June (SE = 0.2). Rainfall data are better characterized by modes and measures of variation than by means. For 134 years, the "modal year" had less than 25 cm of rain, and only 22 years were within 10% of the average. The coefficient of variability for annual rainfall

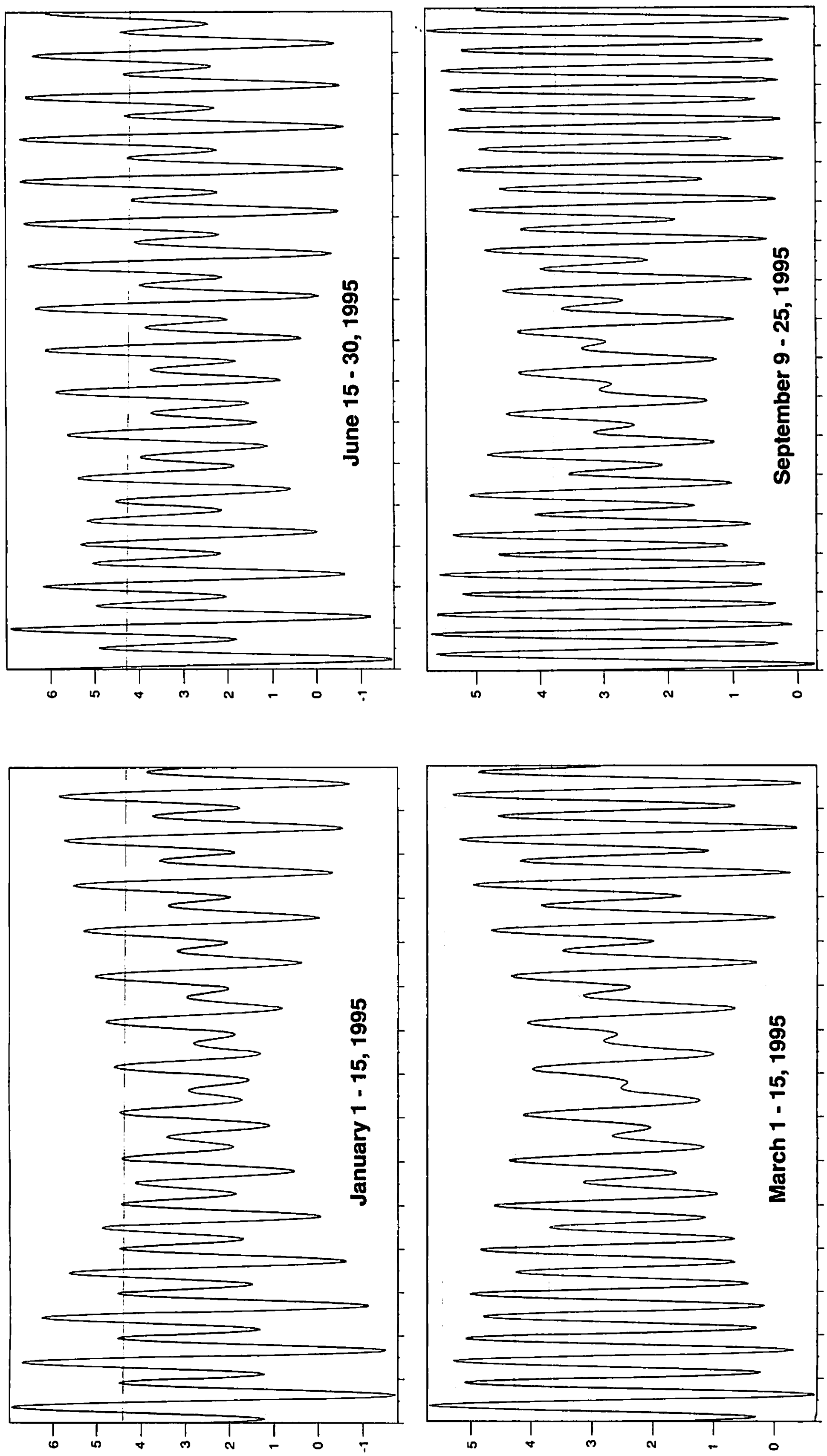
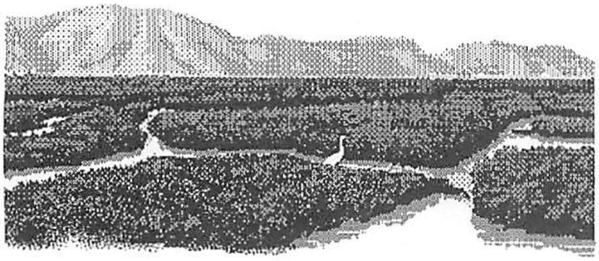


Figure 1.4 Seasonal comparison of tides during daylight (clear) and night (stippled) for San Diego Bay, California. Note that *y*-axis differs for upper and lower graphs.



is 41%; the SE is 0.9; the range is 8–70 cm. Because of these high interannual variations in the timing and in total amount of rainfall, the potential for variations in streamflow is great.

Natural streamflows to Tijuana Estuary are modified by Rodriguez Dam; municipal discharges of imported water to the Tijuana River, mostly in the form of renegade sewage flows from the City of Tijuana (cf. section 3.2); and by agricultural runoff. Streamflow is variable within months, with standard errors as high as 350 million cubic meters; the least variable months are those with low streamflow. Seasonal flows lead to high intra-annual variability; years of low rainfall have little streamflow in the winter and little difference in streamflow from month to month.

Tijuana Estuary is influenced by semidiurnal mixed tides. Approximately 50% of the tidal prism (the average volume of water exchanged by the tides) is attributable to the northern arm (Oneonta Slough) of the estuary (Williams and Swanson 1987). Historic air photos (from 1928 to the present) show that although the main channel has been affected by wave washovers and dune erosion, the smaller channels and tidal creeks have changed little (Zedler et al., 1986). Mean sea level rises about 21 cm/century along the San Diego shore (Flick and Cayan 1985). However, during the 1983 El Niño, the average sea level was 15 cm higher than predicted (R. Flick, personal communication) and storms further raised water levels.

In summary, Tijuana Estuary, like many small wetlands on the Pacific Coast, has high seasonal and interannual variability. It also shares with most other coastal ecosystems a history of hydrologic modification and human encroachment. The combination of natural and anthropogenic disturbances is unique for each estuary and lagoon; hence, no two coastal wetlands are biologically identical. Rather, a spectrum of conditions exists, from less to more disturbed, and the differences provide a general understanding of which species are more tolerant and widespread, and which are more sensitive and more restrictive in their occurrence.

1.2 NATURAL FUNCTIONS AND VALUES

What an estuary does constitutes its many functions. Estuaries are responsible for the primary production (photosynthesis of both vascular plants and algae) of material that fuels food chains. In addition, these systems occur in sheltered areas that are attractive feeding sites for both fish and birds and thus function as refuges and nurseries for selected species. Finally, their emergent wetlands filter sediments and nutrients from the watershed, stabilize shorelines, and serve as buffers when flooding occurs. Those functions the public considers beneficial are its values. Thus, the production of salmon is a value, whereas the release of dihydrogen sulfide (odor of rotten eggs) is not. Neither the functions nor the values of Pacific Coast wetlands have been adequately assessed; however, their role in support of fisheries and the Pacific Flyway has been investigated.

1.2.1 Fisheries Support

The term estuary is almost synonymous with coastal fish and shellfish habitat in the United States, and most of the nation's legislation concerning coastal wetlands and estuaries is geared toward fisheries management. Yet many Pacific coastal estuaries are too small to play a dominant role in fisheries support. Coastal upwelling supports the plankton-based food chain along the Pacific Coast, with relatively minor contributions of detritus from coastal marshes. According to McHugh (1976), estuarine-dependent fisheries accounted for 54% of the Washington catch and 37% of the Oregon catch, but only 3% of the California catch. However, Onuf and Quammen (1985) suggest that the California contribution is diluted from perhaps 7%, because Mexican-caught fish are included in the data on California port landings.

Still, the estuaries provide critical habitat for a number of fish and shellfish species. These include salmon (coho, *Onchorynchus kisutch*; chum, *O. keta*; chinook, *O. tshawytscha*), California halibut (*Paralichthys californicus*), Dungeness crab (*Cancer magister*), Pacific oyster (*Crassostrea gigas*, introduced from Japan), and clams (littleneck clam, *Protothaca staminea*; butter clam, *Saxidomus giganteus*; soft-shelled clam, *Mya arenaria*, introduced from the East Coast). The highest catches are along the Pacific Northwest states. On the basis of their short-term use of estuaries, sockeye salmon (*O. nerka*) and pink salmon (*O. gorbuscha*) were not considered to be estuarine-dependent by Onuf and Quammen (1985), although both starry flounder (*Platichthys stellatus*) and English sole (*Pleuronectes vetulus*) were considered good candidates for estuarine dependency. In California's Mugu Lagoon, shovelnose guitarfish (*Rhinobatos productus*) are summer residents, and in Elkhorn Slough, leopard sharks (*Triakis semifasciata*) and bat rays (*Myliobatis californica*) are caught for commercial use (Onuf and Quammen 1985).

In summarizing the fish support function of Pacific Coast estuaries, Onuf and Quammen (1985, p. 126) concluded, "Wetlands here do not yield food chain support by sheer mass of organic matter produced, as they may elsewhere. Instead, food chain support arises from the use of the wetlands in special ways by the young of a few stocks . . . food chain support is not directly tied to overall primary productivity of the wetland . . . Prey selectivity suggests that quality or availability is more important than amount." Eelgrass (*Zostera marina*) beds in the shallow waters and emergent vegetation along tidal creeks serve as refuges for fishes that are subject to predation (C.A. Simenstad, personal communication).

1.2.2 Pacific Flyway Support

Many types of water-associated birds are strongly linked to wetlands within the Pacific Flyway. The marshes and adjacent habitats serve as migratory stopovers and as breeding and wintering grounds. The most numerous birds are waterfowl, with shorebirds second. Onuf and Quammen (1985) note that most of the Pacific waterfowl winter in California's Central Valley. Along the coast, shorebirds dominate the use of intertidal flats and marshes, although ducks, geese, coots, gulls, and herons are also common. Western sandpipers (*Calidris mauri*), dunlin (*Calidris alpina*), dowitchers (*Limnodromus* spp.), and American avocet (*Recurvirostra americana*) are abundant on mudflats (Quammen 1982, 1984).

Efforts are under way by the Point Reyes Bird Observatory (Stinson Beach, California) to census all wetlands in the Pacific Flyway for the support of shorebirds. Preliminary results indicate that shorebirds concentrate on a few large wetlands, including the Copper and Stikine River deltas in Alaska, the Fraser River delta in British Columbia, Grays Harbor and Willapa Bay in Washington, the Columbia River estuary, and San Francisco Bay, each of which reported more than 100,000 birds. Many other coastal and inland wetlands contribute to the network of stopovers used by shorebirds during spring and fall migrations.

Although the shorebirds, gulls, and herons have no direct commercial value, they provide substantial recreation for a growing number of outdoor activities, especially birdwatching. Nature appreciation has been valued at $1.8 billion/year in California ($129.18 per person, or 27% more than the national average; Duda 1991).

1.2.3 Education and Research

Four Pacific estuaries are included in the National Estuarine Research Reserve program: Padilla Bay in Washington, South Slough in Oregon, and Elkhorn Slough and Tijuana Estuary in California. Each site is responsible for conducting educational programs, interpreting the natural resources for the public (e.g., Silberstein and Campbell 1989) and offering research opportunities (e.g., Zedler et al. 1992).

Additional sites are also important for teaching and research. Nature interpretive centers are appearing in many publicly owned reserves (e.g., Sweetwater Marsh National Wildlife Refuge on San Diego Bay). Some of California's more studied wetland ecosystems are found

in San Diego Bay, Carpinteria Marsh, Mugu Lagoon, Elkhorn Slough, Morro Bay, San Francisco Bay, Bodega Bay, Bolinas Lagoon, and Tomales Bay.

1.2.4 Improvement in Water Quality

Although improvement in water quality is an important function for inland wetlands with downstream water users, the removal of sediments and nutrients from water flowing through Pacific estuaries is often considered less valuable than removal in areas that have downstream users of the flowing water. As saline water is little used (salt production is an exception), the main water-quality concern is eutrophication or pollution of the ocean by nutrients. However, the Pacific Ocean is considered nutrient limited, so eutrophication is rarely an issue. The high variability in streamflows means that enormous quantities of materials flow through estuaries during rare events, reducing both the potential for significant removal and the ability to measure it. Also, the steep slope and narrow continental shelf reduce the potential for accumulation of sediments. Sediment accumulation is not a problem for the open coast, because beaches are often sediment starved. Where sediments are a problem is in the coastal lagoons, where they reduce tidal prisms and tidal action. For example, Mugu Lagoon lost 40% of its low-tide volume during the floods of 1978–1980 (Onuf 1987). Rather than being perceived as an improvement in water quality, accumulation of sediments in coastal wetlands is considered an adverse attribute for resource management. The process is a natural one, but the rates have been greatly accelerated by two modifications: the construction of roadways (e.g., Pacific Coast Highway) across wetlands, which decreases tidal scour and traps inflowing sediments, and increased erosion in coastal watersheds, which results from intensive development and agriculture.

Removal of contaminants and pathogens by coastal wetlands is poorly known, but these issues make headlines in regions where tourism is important to the economy. Certainly, opportunities exist for using coastal freshwater wetlands for wastewater improvement, as shown by the success of the Arcata treatment wetlands at Humboldt Bay in Northern California (Gearheart et al. 1983). Here the secondary effluent from a treatment plant passes through shallow-water habitats that support wetland vegetation and attract waterfowl. Jogging trails and bird-viewing blinds facilitate nature appreciation.

1.3 THE NEED FOR RESTORATION

The need to restore damaged and degraded ecosystems has received national attention. The National Research Council (1992) recommended that a national aquatic ecosystem restoration strategy be developed for the United States. Among the elements proposed were: That goals be established within ecoregions (areas with broadly similar soils, relief, and dominant vegetation), that governmental agencies redesign policies and programs to emphasize restoration, and that innovative measures be used to finance restoration work. A number of specific suggestions were made to move wetland restoration from a trial-and-error process to a predictive science. While no national restoration program has been established, the level of interest has certainly been raised, and the National Science Foundation now has a competitive grants program in conservation and restoration.

1.3.1 Pacific Coast Perspective

Restoration of estuarine wetlands is a high priority in many regions of the Pacific Coast, because so little of these habitats remain: the area of wetlands was never great, and a high percentage of historical wetlands has been lost. According to data cited by NOAA (1990), 90% of California's coastal wetland acreage has been destroyed since settlement. About half the remaining coastal wetlands in California occur in San Francisco Bay. Records compiled by the U.S. Fish and Wildlife Service (Dahl 1990) for coastal plus inland wetlands indicate that Alaska has the nation's lowest rate of wetland loss (<1%), whereas California has the highest (91%); Washington has lost 31%, Oregon 38%, and Hawaii 12%. The combined loss of fresh and saline wetlands is no doubt responsible for losses of avian resources and reduced coastal water quality (especially contaminants). The California coast has experienced a high loss of biodiversity and is undergoing severe development pressure. Restoration efforts are thus badly needed.

After a broad-based review of the status of aquatic ecosystems, the National Research Council (1992) called for 1 million acres (400,000 ha) of wetland to be restored by the year 2010. If no further loss occurs, this is equivalent to a 10% restoration of the nation's historical wetlands acreage. If 10% of the wetland area lost in California were restored, it would add about 455,000 acres (182,000 ha). This would double the current acreage.

Pacific Coast problems and needs are not easily satisfied by policies for resource management that may work elsewhere. Participants in a symposium to determine research needs, held at the 1991 Estuarine Research Federation, agreed that Pacific coastal wetlands have unique features and that management values and methods must be developed with those features in mind (Williams and Zedler 1992). This regional difference was also recognized in a broader review by the National Research Council (1992).

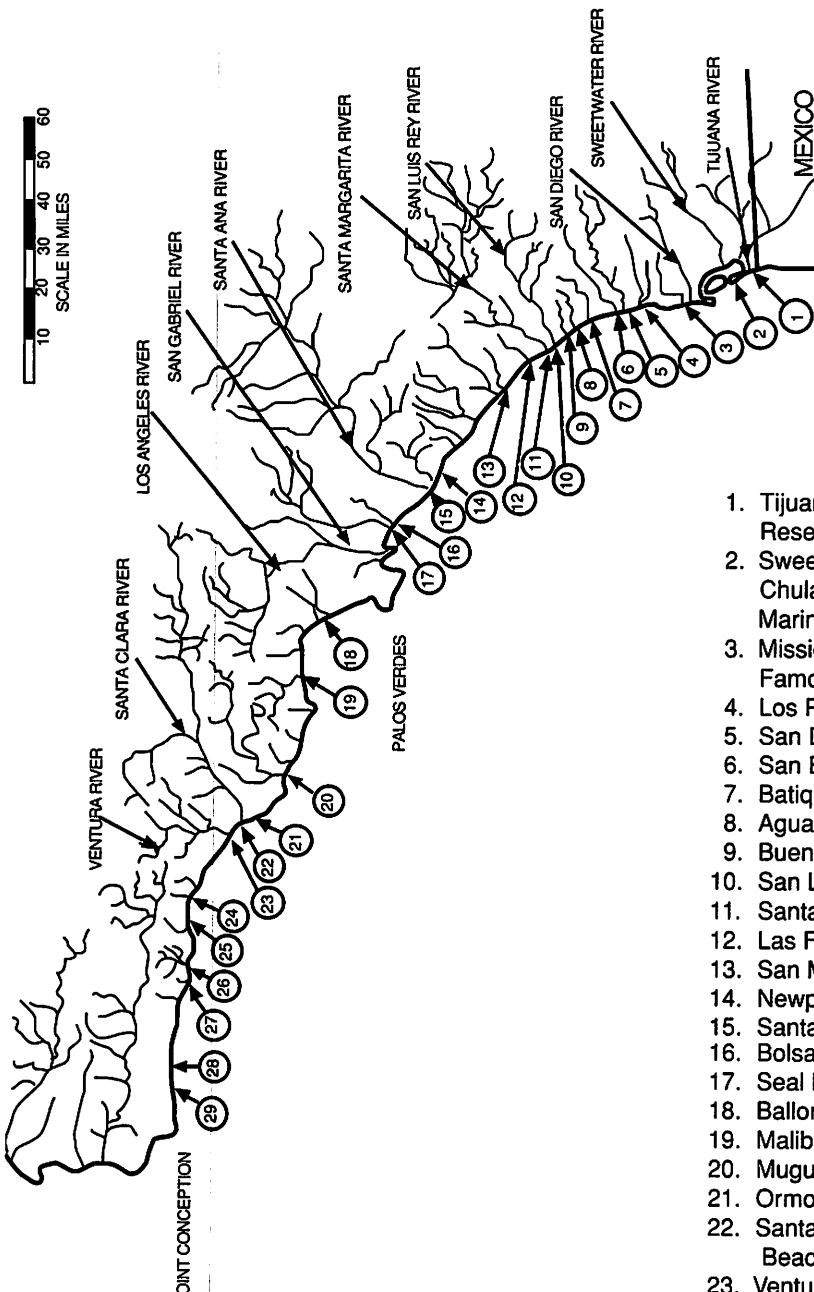
1.3.2 Southern California Perspective

The Southern California coast has many features that make it useful to the analysis of coastal wetland restoration problems and solutions: a high rate of wetland loss, a high and ever-increasing population density, many kinds of impacts of development, and documentation of various efforts to restore damaged wetlands. Historical losses in Southern California's coastal wetlands and ongoing disturbances related to the urban locations of the wetlands have made these ecosystems as threatened as the endangered species they support. Southern California has the nation's most densely populated estuarine drainage areas (San Pedro and Santa Monica Bays, [NOAA 1990]). The region began to grow rapidly after World War II and never stopped. Between 1987 and 1989, approximately 87,000 people moved to San Diego each year. More recently (1991–1994) growth has been approximately 47,000 additional residents annually.

Data (Macdonald 1990) indicate that the resources that have been lost in Southern California coastal wetlands are primarily shallow-water habitats, that is, salt marshes and intertidal flats, rather than the subtidal fish habitats (Table 1.3). Each of the region's 29 remaining coastal wetlands (Fig. 1.5) has many unique features (Purer 1942; Carpelan 1969; Frey et al. 1970; Speth et al. 1970; Browning and Speth 1973; Mudie et al. 1974; Macdonald 1976a, 1976b; Mudie and Browning 1976; Speth et al. 1976; Ferren 1985; Onuf 1987). Management issues are equally diverse; they include concerns about how to maintain regional biodiversity, manage wastewater inflows (including urban runoff), minimize effects of sewage spills, reduce effects of ongoing human activities, prevent algal blooms and fish kills, restore damaged wetlands, and mitigate damages due to future disturbances.

Restoration of Southern California estuaries is a high priority for both the resource agencies and the public. The California Fish and Game Commission has called for a 50% increase in the area of wetlands statewide (State Concurrent Resolution No. 28, Sept. 13, 1979), and a recently formed citizen group, Save California Wetlands, has organized more than 90 nonprofit organizations to begin planning statewide, science-based wetland restoration (M. Hanscom, Save California Wetlands, Coordinator for Southern California, personal communication).

The goal of restoration is to provide self-sustaining ecosystems that closely resemble natural systems in both



1. Tijuana River National Estuarine Research Reserve
2. Sweetwater Marsh National Wildlife Refuge, Chula Vista Wildlife Reserve, South San Diego Marine Reserve
3. Mission Bay Marsh (Kendall-Frost Reserve), Famosa Slough, San Diego River Marsh
4. Los Peñasquitos Lagoon
5. San Dieguito Lagoon
6. San Elijo Lagoon Reserve and Regional Park
7. Batiquitos Lagoon
8. Agua Hedionda Lagoon
9. Buena Vista Lagoon
10. San Luis Rey River Marsh
11. Santa Margarita River Estuary
12. Las Flores Marsh
13. San Mateo Marsh
14. Newport Backbay State Ecological Reserve
15. Santa Ana River Marsh
16. Bolsa Chica Ecological Reserve
17. Seal Beach National Wildlife Refuge
18. Ballona Wetlands
19. Malibu Lagoon
20. Mugu Lagoon
21. Ormond Beach Wetlands
22. Santa Clara River Estuary and McGrath State Beach
23. Ventura River Estuary
24. Carpinteria Salt Marsh Reserve
25. Arroyo Burro Estuary
26. Goleta Slough State Ecological Reserve
27. Deveraux Slough
28. Gaviota Creek Estuary
29. Santa Anita Estuary

Figure 1.5 Location of major Southern California coastal wetlands and rivers

Table 1.3. Habitat Areas and Changes at Tijuana Estuary, San Diego Bay, and Mission Bay, San Diego County, California

	Historic 1856	Recent 1984–87	Area Change	% Change
Salt ponds	0	507	+507	
Intertidal				
salt marsh	1,926	255		
mudflats, sandflats	2,503	407		
Total Intertidal	4,429	662	−3,767	−85.1
Subtidal (rel. to MLLW)				
0 to −1.8 m (6')	3,105	973		
−1.8 to −5.5 m (18')	984	2,318		
below −5.5 m	925	1,727		
Total Subtidal	5,014	5,018	+4	+<0.1

Note: MLLW = Mean lower low water. Areas are ha. Data from Macdonald 1990.

structure and function. However, experience shows that restoration sites do not function as well as natural wetlands (see chapters 2 and 3 and section 5.1). Case studies from British Columbia to the Mexican border indicate a wide range of problems (chapter 2), and detailed investigations in San Diego Bay (Langis et al. 1991, Zedler 1993, Gibson et al. 1994) show that mitigation sites can be functionally impaired for more than a decade after construction. Furthermore, measures to correct the shortcomings of mitigation projects can be lengthy and costly (section 5.1).

Restoring the functions and values of coastal wetlands requires appropriate planning, improved implementation, extensive and intensive assessment, and long-term adaptive management. It is important to consider what habitats were historically present as well as what values are still present at sites selected for construction or restoration. Further efforts are needed to determine what impairs the functioning of restored ecosystems (over the short and long term), and new methods will be required to accelerate the development of critical ecosystem processes.

1.4 THE GOAL OF THIS BOOK

Restoration efforts will always be based on incomplete knowledge of the physical and biological conditions and processes of emergent marshes. We are on a steep learning curve, both because restoration is a relatively new management tool and because so few attempts have been carefully studied. The primary goal of ecological studies of restoration efforts is understanding how ecosystem functions develop and how they compare with naturally functioning systems. Knowledge of natural wetlands helps in setting performance standards, and knowledge of created wetlands helps in determining limiting factors and in developing solutions to problems. With understanding should come greater ability to restore or create naturally functioning wetlands and greater predictability about when and how much natural functioning will develop.

In this book, we review several efforts toward wetland restoration (chapter 2), focusing on the work of PERL. Restoration of tidal flushing, channel habitat, and an endangered plant population are considered in detail. Although our work has been confined to Southern California, our review of projects along the Pacific Coast shows that problems are widespread; each case study provides one or two important lessons for future restoration attempts. On the basis of these and our own experiences, we indicate a number of problems that constrain restoration (chapter 3), including the roles of excess runoff, urbanization, and invasions of exotic plants. Next, we recommend ways to improve planning by taking a regional approach (chapter 4) and to accelerate the development of ecosystems (chapter 5). We then offer suggestions for improving the assessment of restoration sites (chapter 6), including recent applications of high-resolution remote sensing. We conclude with recommendations for managing restoration sites adaptively (chapter 7).

This is not a comprehensive guide to restoration of coastal wetlands. It is an update of findings and scientific perspectives that have developed along with our Southern California research program. We summarize briefly the information that has been published in, or submitted to, the peer-reviewed literature. We provide more details for work that has not been published elsewhere. The information here builds on earlier restoration-oriented guidebooks (Zedler 1984, PERL 1990), but we still consider our research far from complete. In future years, we expect to understand better the problems of wetland restoration and to have more corrective measures. Most of all, we hope to report examples of restored wetlands that are functionally equivalent to natural ones.

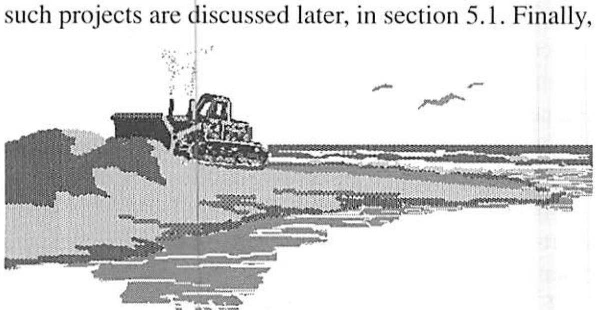
CHAPTER 2

Restoration Efforts on the Pacific Coast

2.1 INTRODUCTION

Wetland restoration is undertaken by resource agencies to improve stocks of selected species or to enhance biodiversity in general. Often restoration is accomplished through mitigation programs that give credit for improving habitat quality or quantity in exchange for allowing damages to other wetlands. This chapter is a review of several Pacific Coast restoration efforts. It is not exhaustive, but the examples illustrate a broad range of issues and problems in the restoration of coastal wetlands, for example, the value of tidal flow, the creation of fish habitat, and the challenge of sustaining populations of annual plants in the salt marsh.

PERL has been directly involved in the projects discussed in sections 2.2–2.4, and both the difficulties of and promising approaches for enhancing biodiversity in Southern California coastal wetlands are discussed. In section 2.2, we compare four wetlands and relate biodiversity to tidal hydrology. This comparison of sites with different degrees of tidal action provides a scientific basis for the common restoration goal of increasing tidal influence. In section 2.3, we evaluate efforts to create tidal channel habitat for fish and benthic invertebrates, which are needed as forage by endangered birds. Many proposals have been developed to create fish habitat in Southern California as mitigation for filling of nearshore waters to expand port facilities and for entrainment of fish larvae by cooling water intakes of power plants. In section 2.4, we describe problems encountered in reestablishing an endangered plant. Attempts to restore cordgrass vegetation for nesting by light-footed clapper rails have encountered difficulties; efforts to improve such projects are discussed later, in section 5.1. Finally, in section 2.5, several brief case studies from California to British Columbia are presented. Each indicates a problem or concern that serves as a lesson for future restoration or habitat-construction projects.



2.2 MAINTAINING TIDAL FLOWS IN SOUTHERN CALIFORNIA LAGOONS

John Boland and Joy Zedler

Urban development has reduced the size of Southern California wetlands, and roadways have dissected many of them into two or three basins connected only by narrow channels. Several bodies of water that were once open to the ocean are now often closed to tidal flow (California State Coastal Conservancy 1989). In this condition, they are usually called lagoons and often are considered "degraded." Current restoration plans call for restoring and maintaining full tidal flushing at San Dieguito Lagoon (MEC 1991), Batiquitos Lagoon (City of Carlsbad and U.S. Army Corps of Engineers 1990), and the southern arm of Tijuana Estuary (Entrix et al. 1991). The San Dieguito and Batiquitos lagoon projects have been designed to mitigate losses to nearshore fish populations; funding has not been determined for the Tijuana Estuary Tidal Restoration Plan. Additional plans have been developed to improve tidal flows to Ballona Wetland just north of Los Angeles International Airport. Most of this wetland was excavated to create Marina del Rey. The site is virtually nontidal; it is connected to the ocean by culverts that have one-way flap gates to allow freshwater outflow but little tidal inflow (Boland and Zedler 1991).

Although restoration of tidal flows is considered the best way to enhance many of these lagoons, the cause-effect relationships between tidal inflow and biodiversity have not been fully evaluated. Here we use data from four coastal wetlands to compare the biota of sites with different tidal-flushing regimes (Fig. 1.5, sites 18, 6, 4, and 1). These patterns of occurrence in relation to tidal flow suggest cause-effect hypotheses, which can be tested by studying lagoons before and after restoration of tidal flows. Such studies are needed to predict more precisely the benefits of increased tidal flow and, for mitigation projects, to establish mitigation credits associated with different degrees of tidal enhancement.

2.2.1 Differences in Tidal Flushing

The four study sites are subdivided into three hydrologic types according to the nature of the ocean inlet. These conditions represent different points along a spectrum that characterizes bodies of water along the Southern California coast (see Ferren et al. 1995 for a more detailed hydrologic classification scheme):

1. Fully tidal, inlet nearly always open (e.g., Tijuana Estuary)
2. Inlet frequently closed (e.g., San Dieguito Lagoon and Los Peñasquitos Lagoon)
3. Inlet with a berm, weir, or tide gate allowing water to flow out but excluding tidal inflow (e.g., Ballona Wetland)

The northern arm of Tijuana Estuary is a fully tidal system; this part of the estuary has received more study (Nordby and Zedler 1991; Zedler, Nordby, and Griswold 1990; Zedler, Nordby, and Kus 1992) than the southern, less tidally influenced arm (Entrix et al. 1991). Reduced tidal influence in the southern arm is a function of sedimentation from the watershed and dune washover events. Both the area and the lengths of channels have been reduced, and much of the salt marsh has accumulated enough sediment to raise the marsh above tidal influence.

Examples of frequently closed lagoons are Los Peñasquitos Lagoon (Covin 1987; Greenwald and Britton 1987; Nordby and Covin 1988; Nordby 1989, 1990a; Nordby and Zedler 1991; and Boland 1991, 1992a, 1993a, 1993b) and San Elijo Lagoon (Nordby 1990b; Boland 1992b, 1993c, 1994). Inlets are typically open during winter after rainfall that is sufficient for runoff to break through the sand berm at the inlet. Hence, lagoons are susceptible to freshwater inundation in winter and evaporation and hypersalinity in summer. Both the timing and the duration of closures are highly variable (Webb 1989).

Ballona Wetland is an example of a wetland where tides are excluded by a tide gate (PERL 1989, Boland and Zedler 1991, PERL unpublished data). Other wetlands in the region that lack tidal influence have inlets with a weir (Buena Vista Lagoon) or persistent buildup of cobbles and sand, which form a high berm (Batiquitos Lagoon).

2.2.2 Channel Characteristics and Aquatic Organisms

The salinity of channel water influences the abundance of species within the channels. Most estuarine fish and invertebrates are species that thrive under marine conditions, and their abundances are thought to decline under extended periods with water salinity less than 10 ppt, that is, less than one-third seawater (e.g., California halibut *Paralichthys californicus*; Baczkowski 1992). When Tijuana Estuary is fully tidal, the average salinity of the surface water in the tidal channel fluctuates little, and during dry weather, it averages 32 ppt (Fig. 2.1). At the other sites, salinities are likely to be more stressful to organisms in the channel. At the frequently closed lagoons, conditions are less predictable: San Elijo Lagoon is generally brackish, whereas Los Peñasquitos Lagoon can be brackish or hypersaline. At San Elijo Lagoon, freshwater inflows have exceeded evaporation, whereas at Los Peñasquitos Lagoon, the reverse has been true. At Ballona Wetland, an upstream dewatering project released fresh water into the creek and reduced channel salinities during 3 months in late summer. Thus, the condition of the ocean inlet has a major influence on water salinity, but the effect depends on the amount of streamflow.

Poorly flushed channels are suitable places for rapid growth of macroalgae and the vascular plant *Ruppia maritima*. Plant biomass increases during summer, and then during autumn the plants begin to die. The decomposers of these plants become extremely abundant and metabolically deplete the amount of dissolved oxygen in the water column, particularly at night.

Oxygen concentrations affect the abundances of channel organisms. Low concentrations can cause extensive mortality of fish and invertebrates. In October 1987, 10 species and an estimated 6,200 fish were killed at Los Peñasquitos Lagoon during one of these events. The counts done after that kill illustrate not only the magnitude of the problem but also the nature of the channel biota: about 5,000 topsmelt, *Atherinops affinis*; 515 diamond turbot, *Hypsopsetta guttulata*; 500 deepbody anchovy, *Anchoa compressa;* 134 California halibut, *Paralichthys californicus;* 11 Pacific staghorn sculpin, *Leptocottus armatus*; 4 barred sand bass, *Paralabrax nebulifer;* 2 California corbina, *Menticirrhus undulatus*; 2 shiner surfperch, *Cymatogaster aggregata;* 2 opaleye, *Girella nigricans;* and 1 yellowfin goby, *Acanthogobius flavimanus* (C. Nordby, unpublished data). In fully tidal systems, oxygen concentrations are consistently above lethal levels, whereas in frequently closed estuaries, concentrations often decrease to less than 4 mg/L, particularly during late summer and fall (Fig. 2.2).

The channel sediments of frequently closed estuaries are usually very fine, highly organic, and anaerobic, whereas the sediments at fully tidal estuaries have more sand and are more readily aerated at low tide. The oxygen content of the sediment influences the organisms living in the sediments, and many species are intolerant of frequent anoxia. The benthic invertebrate community at Tijuana Estuary had more species, particularly more large, long-lived species (e.g., bivalve molluscs, *Callianassa californiensis*), than the other estuaries (Table 2.1). Small, short-lived polychaete worms were more abundant at the other three sites.

Fish were more abundant at Tijuana Estuary than at

Table 2.1. Numbers of Individuals of Benthic Macroinvertebrates Collected at Four Southern California Coastal Wetlands

Taxon	Numbers at BW	TJE	LPL	SEL
Sipunculid worms				
Themiste sp.	0	17	0	0
Echinoid echinoderms				
Dendraster excentricus	0	6	3	0
Nemertean worms	1	93	3	0
Polychaete worms				
Capitellidae	69	814	399	29
Spionidae	41			
Boccardia spp.	0	68 (5)	183 (4)	0
Polydora cornuta	0	124	18	1
Polydora ligni	0	143	92	0
Polydora spp.	0	63 (2)	210 (2)	2,631 (2)
Spiophanes missionensis	0	117	0	0
Opheliidae				
Euzonus mucronata	0	0	162	0
Other taxa combined	11 (3)	437	161	46 (2)
Bivalve molluscs				
Tagelus californianus	1	797	40	7
Protothaca staminea	0	554	4	0
Macoma nasuta	1	221	6	0
Laevicardium substriatum	0	30	8	0
Spisula sp.	0	0	17	0
Other species combined	1 (1)	227	17	0
Other molluscs				
Cerithidea californica	55	0	0	0
Assiminea	1	0	0	37
Lacuna	0	0	0	1
Cylichnella	0	0	0	1
Decapod crustaceans				
Callianassa californiensis	0	234	3	0
Other crustaceans				
Amphipods	57	0	0	2,104
Isopods	0	0	0	1
Insecta				
Fly larvae	0	0	0	145
Phoronida				
Phoronis sp.	0	1	114	0
Total number of taxa	12	22	21	13
Total number of individuals	238	3,946	1,440	5,003
Mean density (individuals/m^2)	1,253	752	377	1,573
Total sampling area (m^2)	0.19	5.25	3.82	3.18
Total number of stations (seasons × sites)	4 × 6	11 × 3	8 × 3	5 × 4
Dates	1989–1990	1986–1988	1987–1988	1991–1993

Note: Numbers in parentheses are numbers of species. BW = Ballona Wetland, TJE = Tijuana Estuary, LPL = Los Peñasquitos Lagoon, SEL = San Elijo Lagoon.

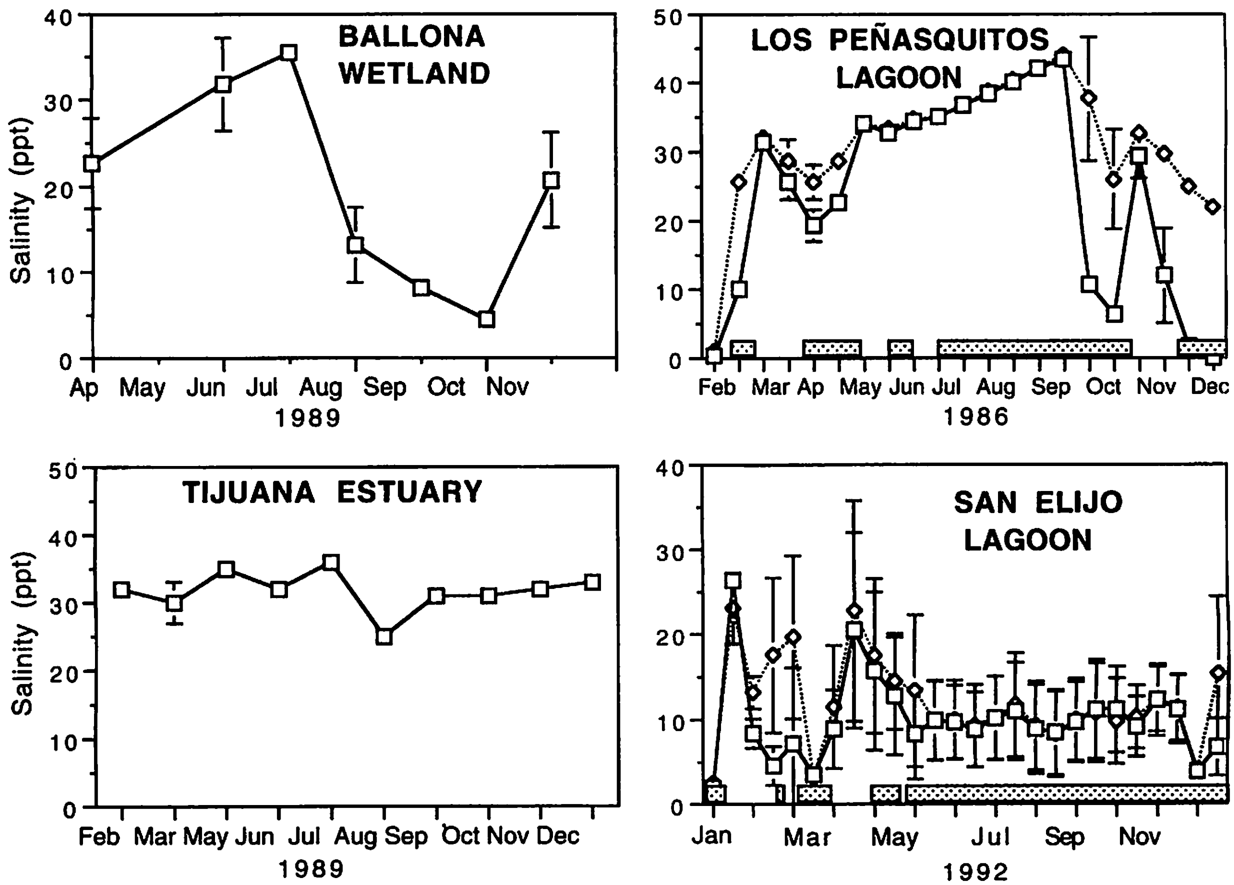


Figure 2.1 Salinities of channel waters at four Southern California estuaries. Surface salinities only are given for Ballona Wetland and Tijuana Estuary. Both surface and bottom salinities and timing of mouth closure are shown for Los Peñasquitos Lagoon and San Elijo Lagoon. Error bars are ± 1 SD. Number of sample sites: Ballona Wetland, six; Tijuana Estuary, four; Los Peñasquitos Lagoon, three; San Elijo Lagoon, five. ☐ mouth closure, ...◇...... bottom salinity, —☐— surface salinity.

the other sites (Table 2.2). Both the number of species and the density of individuals were greatest at Tijuana Estuary. The community compositions at Tijuana Estuary, Los Peñasquitos Lagoon, and San Elijo Lagoon were similar; arrow goby, topsmelt, California killifish, longjaw mudsucker, and staghorn sculpin were the most abundant species. At Ballona Wetland the less salt-tolerant mosquitofish was the most abundant species. (Note that the fish and invertebrate data sets have different degrees of sampling intensity and span different years. Their usefulness in making comparisons is limited.)

2.2.3 Relationship of Salt Marsh Vegetation to Tidal Influence

The composition of plant communities differs among the lagoons compared here (Table 2.3). The salt marsh at Tijuana Estuary is richest in species. Three species found here are not found at Ballona Wetland, Los Peñasquitos Lagoon, and San Elijo Lagoon: *Batis maritima*, *Limonium californicum*, and *Spartina foliosa*. At Ballona Wetland and Los Peñasquitos Lagoon the salt marshes are species-poor, and they are essentially monotypes of *Salicornia virginica*. San Elijo Lagoon is similar to Los Peñasquitos Lagoon, except the relative abundance of *Salicornia virginica* is reduced, and brackish marsh species have encroached on the salt marsh. San Elijo Lagoon is usually flooded, and areas that are mudflat would likely support more extensive stands of *Salicornia virginica* if the inundation period were reduced. In fact, *Salicornia virginica* invaded the mudflats when the ocean inlet was kept open for several months in 1994 (J. Boland, personal observation).

In general, Southern California salt marshes with a long history of good tidal flushing tend to have more native salt marsh plant species than marshes with inlets that close (PERL 1990). The elimination of many halophytic plant populations appears to be due to extreme hypersalinity and drought that can accompany inlet closure. We found heavy mortality of *Salicornia bigelovii*, *Suaeda esteroa*, and *Spartina foliosa* in 1984, when Tijuana Estuary was closed to tidal flushing and marsh soils developed salinities of 104 ppt by late summer (Zedler et al. 1992). Similar declines occurred in a dry, nontidal (diked) portion of Estero de Punta Banda, Baja California, that same year (Ibarra-Obando and Poumian-Tapia 1991). Los Peñasquitos Lagoon had a luxuriant population of *S. foliosa* in the late 1930s (Purer 1942). However, the lagoon was closed to tidal flushing for 8

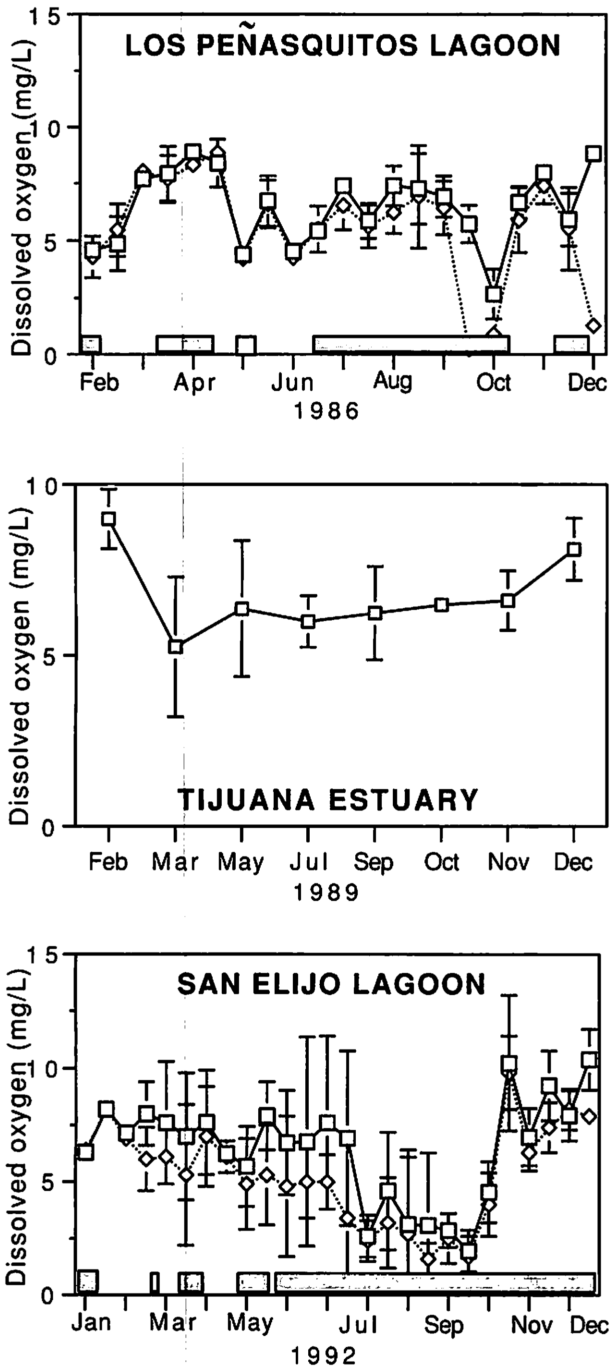


Figure 2.2 Dissolved oxygen concentration of channel waters at three Southern California estuaries. Surface oxygen concentrations only are given for Tijuana Estuary. Both surface and bottom oxygen concentrations and timing of mouth closure are shown for Los Peñasquitos and San Elijo lagoons. Error bars are ± 1 SD. Number of sample sites: Tijuana Estuary, four; Los Peñasquitos Lagoon, four; San Elijo Lagoon, five. ☐ mouth closure, ·····◇····· bottom oxygen concentration, —☐— surface oxygen concentration.

years (December 1958 to 1966), and by 1968 *Spartina foliosa* had disappeared from the area completely (Bradshaw 1968).

Soil salinities in the salt marshes have been sampled in summer in these coastal marshes; they are generally high at Tijuana Estuary and low at the other sites (Fig. 2.3). Maximal heights of *Salicornia viginica* are short at Tijuana Estuary and taller at the other sites (Fig. 2.3). There is a strong pattern of short plant canopies occurring where soils are more saline. However, non-salt marsh vegetation invades where soil salinities remain low for too long (see section 3.4.4).

2.2.4 Resident Salt Marsh Birds

Two endangered bird species breed in the salt marshes; of these, Belding's Savannah sparrow, *Passerculus sandwichensis beldingi*, is the more common. Eight pairs of this species bred at Ballona Wetland, 156 at Los Peñasquitos Lagoon (Boland and Elrod 1993), 30 at San Elijo Lagoon (R. Patton, personal communication), and 144 in the northern arm of Tijuana Estuary. These data suggest that sparrows do well in impounded and fully tidal wetlands but poorly in areas where tides are excluded.

The light-footed clapper rail nests almost exclusively in fully tidal estuaries. The rail was abundant in Tijuana Estuary (85 breeding pairs in 1993), not found in Los Peñasquitos Lagoon and Ballona Wetland, and rare in San Elijo Lagoon (approximately 5 breeding pairs). Los Peñasquitos Lagoon and San Elijo Lagoon once supported large populations of rails, but persistent flooding contributed to making these marshes unsuitable for nesting (Zembel and Massey 1983).

2.2.5 Summary of Patterns Associated with Tidal Influence

These comparisons of three tidal regimes suggest strong correlations between hydrology and biodiversity, with more species present in fully tidal sites (Table 2.4). Salt marsh plants, benthic invertebrates, and fish appeared to be more species-rich in the fully tidal Tijuana Estuary than elsewhere. This is probably due to the narrower range of environmental conditions in fully tidal systems. Estuaries with open inlets are less likely to experience environmental extremes. The three most important extremes appear to be (1) events that produce low oxygen concentrations, which lead to deaths of channel organisms, particularly fish and invertebrates; (2) long-term flooding of the salt marsh by low-salinity water that makes salt marshes unsuitable for breeding birds and may eliminate salt marsh plants that are less tolerant of anoxic soils; and (3) drought and hypersalinity after mouth closure, which eliminate fish, invertebrates, and the less salt-tolerant halophytes (e.g., *Spartina foliosa*).

If the correlation between tidal flushing and biodiversity is a causal relationship, then increasing tidal

Table 2.2. Total Number of Fishes Collected at the Four Southern California Coastal Wetlands

		Numbers at			
Taxon	**Common Nane**	**BW**	**TJE**	**LPL**	**SEL**
Acanthogobius flavimanus	Yellowfin goby	0	0	0	1
Anchoa compressa	Deepbody anchovy	0	11	67	6
Artedius sp.	Sculpin	0	2	0	0
Atherinops affinis	Topsmelt	77	15,437	1,875	1,744
Clevelandia ios	Arrow goby	10	60,097	816	0
Cyprinus carpio	Carp	0	0	0	133
Engraulis mordax	Northern anchovy	0	0	0	34
Fundulus parvipinnis	California killifish	8	2,367	107	30
Gambusia affinis	Mosquitofish	779	0	937	235
Gillichthys mirabilis	Longjaw mudsucker	16	275	877	2
Girella nigricans	Opaleye	0	82	0	0
Hypsoblennius gentilis	Bay blenny	0	4	0	0
Hypsoblennius gilberti	Rockpool blenny	0	1	0	0
Hypsoblennius jenkinsi	Mussel blenny	0	1	0	0
Hypsopsetta guttulata	Diamond turbot	0	83	14	0
Ilypnus gilberti	Cheekspot goby	0	50	22	0
Lepidogobius lepidus	Bay goby	0	0	9	0
Leptocottus armatus	Staghorn sculpin	0	1,431	346	3
Mugil cephalus	Striped mullet	0	5	3	1
Paralabrax clathratus	Kelp bass	0	12	0	0
Paralichthys californicus	California halibut	0	283	12	4
Plueronichthys ritteri	Spotted turbot	0	4	0	0
Quietula y-cauda	Shadow goby	0	3	0	3
Rhinobatos productus	Shovelnose guitarfish	0	2	0	0
Seriphus politus	Queenfish	0	1	0	0
Syngnathus leptorhynchus	Bay pipefish	0	14	2	0
Total number of species		5	21	13	12
Total number of individuals		890	80,165	5,087	2,196
Mean density (individuals/m^2)		5	17	3	1
Total sampling effort (cumulative area in m^2)		192	4,795	1,985	2,000
No. of stations (seasons × sites)		4×6	12×4	8×3	5×4
Dates		1990–1991	1986–1988	1987–1988	1991–1993

flows from frequently nontidal to fully tidal should allow more species to use lagoons that are kept open to the ocean, provided that periods of anoxia and extremes of salinity are either prevented or kept brief.

2.2.6 Effects of Improving Tidal Flushing at Los Peñasquitos Lagoon

Several dredging operations at the ocean inlet between 1990 and 1994 increased the duration of tidal flushing to Los Peñasquitos Lagoon from 23% of the time in 1986 to 95% in 1992. A comparison of data collected before and after tidal flows were improved helps test the linkage between hydrology and biology.

With improved tidal flow, three aspects of Los Peñasquitos Lagoon became more similar to Tijuana Estuary. First, salinities of the channel waters during 1992 (Fig. 2.4) were less variable than those in 1986 (Fig. 2.1). In 1992, channels did not become hypersaline during summer and did not remain low for long periods during rain events. Second, the dissolved oxygen concentrations measured during 1992 were also much higher. The average concentrations on the bottom of the channel never fell below 2.5 mg/L during 1992 (Fig. 2.4), whereas this occurred three times in 1986. Finally, more benthic invertebrate species and individuals were present in September 1993 than in September 1989 (Table 2.5).

However, neither the vascular plant communities nor the general structure of the fish assemblage of Los Peñasquitos Lagoon changed greatly after improvement in tidal flow (Tables 2.6 and 2.7). These communities may require more time to change, or the structure of the lagoon may play a role. Species cannot reinvade a coastal wetland without opportunities for establishment and refuges for them to persist. A fully vegetated salt marsh

Table 2.3. Dominant Salt Marsh Vascular Plants at Four Southern California Coastal Wetlands

	Percent at			
Taxon	**BW**	**TJE**	**LPL**	**SEL**
Batis maritima	0	11	0	0
Cressa truxillensis	6	1	2	9
Distichlis spicata	7	4	2	10
Frankenia grandifolia	2	4	15	10
Jaumea carnosa	1	7	1	15
Lasthenia glabrata	0	0	1	1
Limonium californicum	0	1	0	0
Monanthochloe littoralis	0	8	0	1
Salicornia subterminalis	1	12	1	8
Salicornia virginica	83	31	80	46
Spartina foliosa	0	21	0	0
Total number of species	6	10	7	8
Total number of quadrats	288	277	259	107
Total length of transects (m)	1,475	1,435	1,330	660

Note: BW = Ballona Wetland, TJE = Tijuana Estuary, LPL = Los Peñasquitos Lagoon, SEL = San Elijo Lagoon.

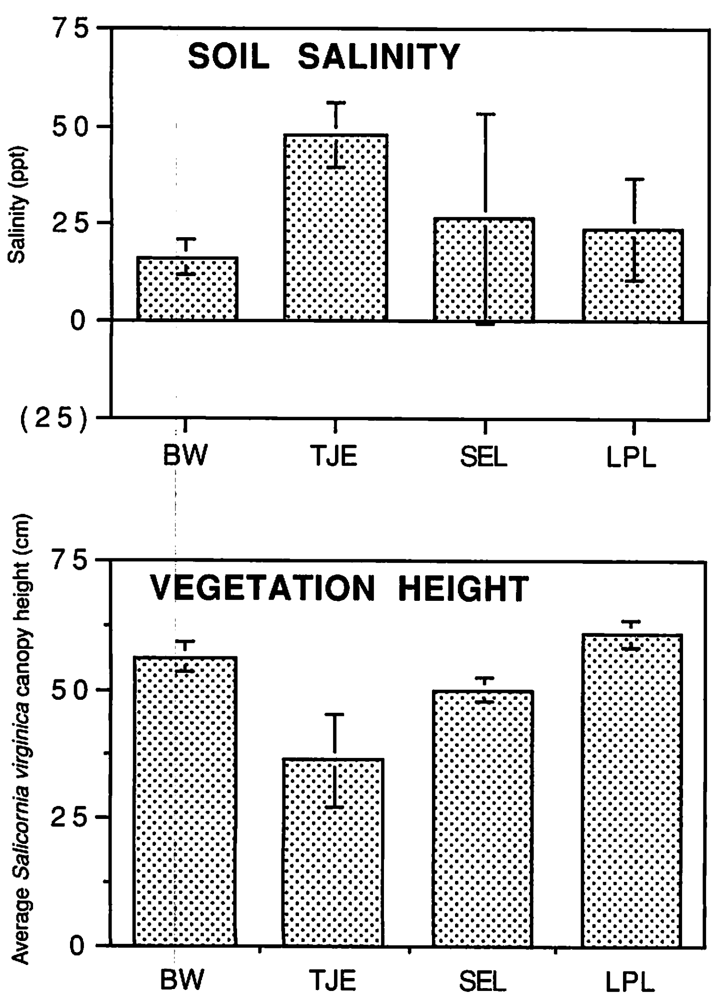


Figure 2.3 Average soil salinities and *Salicornia virginica* canopy heights in the salt marshes at four Southern California estuaries. Error bars for soil salinities are ± 1 SD. Number of sample sites: BW = 26, TJE = 18, SEL = 19, LPL = 29. Error bars for canopy heights are ± 1 SE. Number of sample sites: BW = 27, SEL = 51, LPL = 23, BW = Ballona Wetland, TJE = Tijuana Estuary, SEL = San Elijo Lagoon, LPL = Los Peñasquitos Lagoon.

Table 2.4 Physical, Chemical, and Biological Characteristics of Four Estuaries in Southern California

Characteristic	Site: Ballona Wetland (Tides Excluded)	Tijuana Estuary (Fully Tidal)	Los Peñasquitos and San Elijo Lagoons (Frequently Nontidal)
General			
Nature of mouth	Outflows only	Always open	Seasonally open
Fate of freshwater inflows during:			
Winter/spring	Out mouth	Out mouth	Out mouth
Summer/fall	Out mouth	Out mouth	Impounded
Channels			
Salinities	Variable	Marine	Variable
Oxygen concentration	No data	High	Often low
Dominant plants	*Enteromorpha*	(Phytoplankton)	Phytoplankton, *Ruppia*
Sediment anaerobic depth	At surface	Deep	At surface
No. of invertebrate species, individuals	Few	Many	Few
No. of fish species, individuals	Few	Many	Few
Salt Marshes			
Soil salinities	Low	High	Moderate
No. of plant species	Low	High	Low
Salicornia maximal height	Tall	Short	Tall
Breeding birds	(BSS)	CLT, CR, BSS	(CLT), (CR), BSS

Note: Organisms listed in parentheses are not abundant. BSS = Belding's Savannah sparrow, CLT = California least tern, CR = clapper rail.

offers few open spaces for seedling establishment, especially where the overstory canopy is tall and dense.

We think that fish assemblages are affected by tidal flushing, but that Los Peñasquitos Lagoon has too much freshwater influence even when it is kept open to tidal flows. The fish assemblage at Los Peñasquitos Lagoon is adversely affected by winter population crashes that are not characteristic of Tijuana Estuary (cf. Fig. 7 in Nordby and Zedler 1991). The winter crashes may prevent the fish community from developing fully. At Los Peñasquitos Lagoon, the number and sizes of refuges available to fish during rain-induced floods may be inadequate. Refuges are needed to prevent low-salinity flood waters from killing fish and to keep strong currents from washing fish out to sea. In Los Peñasquitos Lagoon, both main channels carry flood flows to the ocean, and the salinity of virtually the entire lagoon is reduced. In contrast, the northern arm of Tijuana Estuary acts as a refuge, at least during lesser floods. Flood waters rush down the river and out the mouth. The smaller area of tidal influence and the simpler channel system of Los Peñasquitos Lagoon may preclude the development or maintenance of fish assemblages as rich as those that occur at Tijuana Estuary, even if tidal flows were fully restored.

2.2.7 Conclusion

Comparison of the four coastal wetlands that differ in degree of tidal influence suggests that hydrology and biology are strongly related and that fully tidal systems are more diverse biologically and less variable environmentally than frequently closed lagoons.

The changes that occurred at Los Peñasquitos Lagoon show that increased tidal flow made it more like Tijuana Estuary and that rapid improvements occurred in salinity, dissolved oxygen concentration, and benthic invertebrate assemblages. The lack of change in the composition of fish and salt marsh plants suggests that these communities need more time to change or that the structure of the lagoon restricts the number of refuges needed by fish or the opportunities for establishment of vascular plants.

Full tidal flushing is a desirable restoration goal for formerly tidal coastal wetlands in Southern California. With full tidal flushing, Ballona Wetland, Los Peñasquitos Lagoon, and San Elijo Lagoon could

Table 2.5. Abundances of Benthic Macroinvertebrates Collected at Los Peñasquitos Lagoon During September 1989–1993

Taxon	Number			
	1989	1991	1992	1993
Polychaete worms				
Capitellidae	230	28	655	1,534
Spionidae				
Polydora nuchalis	13	36	280	310
Polydora sp.	137	5	1	2
Cirratulidae	9	0	0	0
Opheliidae				
Armandia brevis	0	0	7	19
Glyceridae				
Hemipodus borealis	0	0	0	3
Bivalve molluscs				
Tagelus californianus	18	0	25	209
Protothaca staminea	0	0	2	29
Musculista senhousia	0	0	4	54
Tellina sp.	0	0	1	22
Laevicardium sp.	0	0	0	37
Macoma sp.	0	0	0	9
Other molluscs				
Cylichnella culcitella	40	0	3	18
Cerithidea californica	0	0	85	55
Navanax sp.	0	0	1	0
Bulla gouldiana	0	0	8	10
Alderia modesta	0	0	5	0
Assimenia californica	47	0	0	0
Crustaceans				
Orchestia traskiana	30	552	59	359
Hemigrapsus oregonensis	0	1	11	4
Unknown amphipod	0	0	2	0
Insects				
Fly larvae	2	0	7	0
Trichocorixia reticulata	1	15	0	0
Miscellaneous				
Phoronida	137	0	1	224
Nemertea	0	0	0	4
Asteroidea	0	0	0	1
Total number of taxa	11	6	18	19
Total number of individuals	664	637	1,157	2,903

Note: No data were collected during September 1990. The mouth of the lagoon was closed in 1989 and open during 1991, 1992, and 1993.

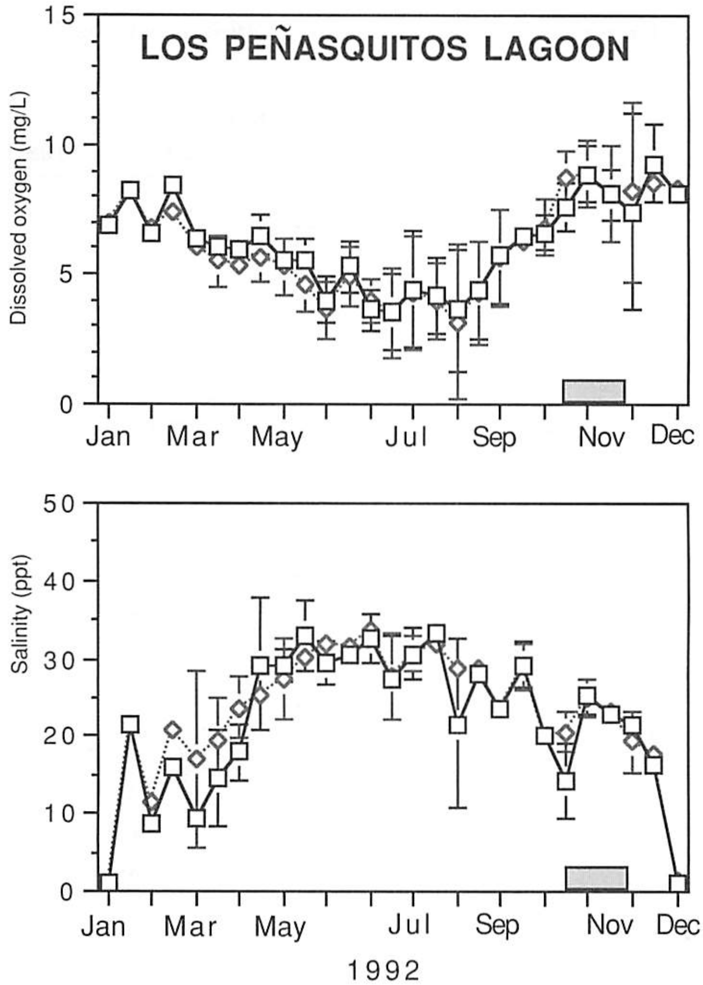
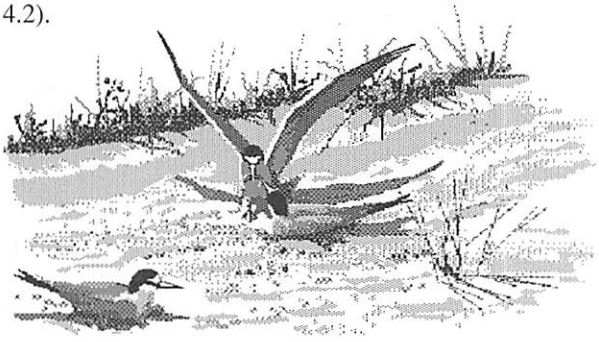


Figure 2.4 Salinity and dissolved oxygen concentrations of surface and bottom channel water and timing of mouth closure at Los Peñasquitos Lagoon during 1992. Error bars are ± 1 SD (n = 3 sites). ■ mouth closure, —□— surface, ····◇···· bottom.

provide more habitat for native vascular plants, and support more endangered birds that reside in the salt marsh and more native marine invertebrates and fishes. However, some sites may support other valued species and have alternative values that would be lost under full tidal flow. For example, many waterfowl use impounded waters as stopovers during migration, and some waders prefer to nest at nontidal water bodies (cf. section 5.3). Individual estuaries should be assessed carefully before restoration of full tidal flow is advocated, and a strategy should be developed to coordinate restoration objectives within the Southern California region (section 4.2).



2.3 CREATING FORAGING HABITAT FOR THE CALIFORNIA LEAST TERN AND LIGHT-FOOTED CLAPPER RAIL

Bruce Nyden and Joy Zedler

As part of a mitigation project under the Endangered Species Act, the California Department of Transportation (Caltrans) was required to create foraging habitat for two endangered species of birds. The California least tern (*Sterna albifrons browni*) requires deepwater habitat for feeding on small fishes; the light-footed clapper rail requires tidal creeks for foraging on crabs and other macroinvertebrates.

Channels were constructed at the Connector Marsh within the Sweetwater Marsh National Wildlife Refuge in 1984. The constructed channels must provide support for at least 75% of the native species and 75% of the individuals of native species found in the reference channels (U.S. Fish and Wildlife Service 1988). In 1989, PERL began comparisons of the fish and invertebrate assemblages of constructed and natural channels. A sampling program to determine water quality was developed to help

Table 2.6. Abundances of Fishes Collected at Los Peñasquitos Lagoon During June 1989–1993

Species	Common Name	Number 1989	1991	1992	1993
Acanthogobius flavimanus	Yellowfin goby	6	0	0	3
Anchoa compressa	Deepbody anchovy	19	0	0	13
Atherinops affinis	Topsmelt	4,822	1,144	222	858
Clevelandia ios	Arrow goby	88	13	1	2
Engraulis mordax	Northern anchovy	0	3	1	0
Fundulus parvipinnis	California killifish	0	9	38	2
Gambusia affinis	Mosquitofish	101	0	0	1
Gillichthys mirabilis	Longjaw mudsucker	96	99	108	70
Girella nigricans	Opaleye	0	0	1	0
Hypsopsetta guttulata	Diamond turbot	0	0	4	0
Ilypnus gilberti	Cheekspot goby	3	0	0	0
Lepidogobius lepidus	Bay goby	0	11	0	0
Leptocottus armatus	Staghorn sculpin	46	0	13	15
Paralichthys californicus	California halibut	0	1	0	2
Plueronichthys ritteri	Spotted turbot	0	1	0	0
Quietula y-cauda	Shadow goby	0	4	0	0
Syngnathus leptorhynchus	Bay pipefish	0	0	0	6
Total number of individuals		5,241	1,282	388	972
Total number of species		8	9	8	10

Note: No data were collected during June 1990. The mouth of the lagoon was closed in 1989 and 1991 and open in 1992 and 1993.

explain similarities and differences between the biota in natural and constructed channels. The monitoring program assists Caltrans, the U.S. Army Corps of Engineers, and the U.S. Fish and Wildlife Service in determining whether constructed channels meet the mitigation requirements (U.S. Fish and Wildlife Service 1988; Table 2.8).

To determine whether the mitigation criteria were met, we used individual-year data to calculate percentages. We listed the native species that use the natural channel sites and determined which of these species also occurred at the constructed channel sites. Note that with this procedure, the percentage of species present in constructed channels cannot exceed 100%. Next, we calculated the average density of the native species in the constructed channels and expressed that number as a percentage of the average density of native species in the natural channels. In this second calculation, the percentage can exceed 100%.

2.3.1 Fish Assemblage

Fish were sampled periodically at six stations, four stations in the constructed marshes and two in the natural marsh (Fig. 2.5). At each station, blocking nets were used to create a sampling area approximately 10 m long. A beach seine was repeatedly pulled through the sample area until it appeared that all the fish had been caught. Through 1991, baited minnow traps were also used to compare the abundances of exotic fish, particularly yellowfin goby (*Acanthogobius flavimanus*). These are the largest fish caught in the Sweetwater channels; many specimens are 10–15 cm long. Yellowfin goby are generalist predators and likely affect the abundance of native species.

The fish assemblages in the natural and constructed sites were slightly different: the assemblage in the natural channel was dominated primarily by arrow goby (*Clevelandia ios*) and California killifish (*Fundulus parvipinnis*), whereas the constructed channel was dominated by topsmelt (*Atherinops affinis*) and California killifish. This difference may encourage the least tern to forage in the constructed channels, because *Atherinops*

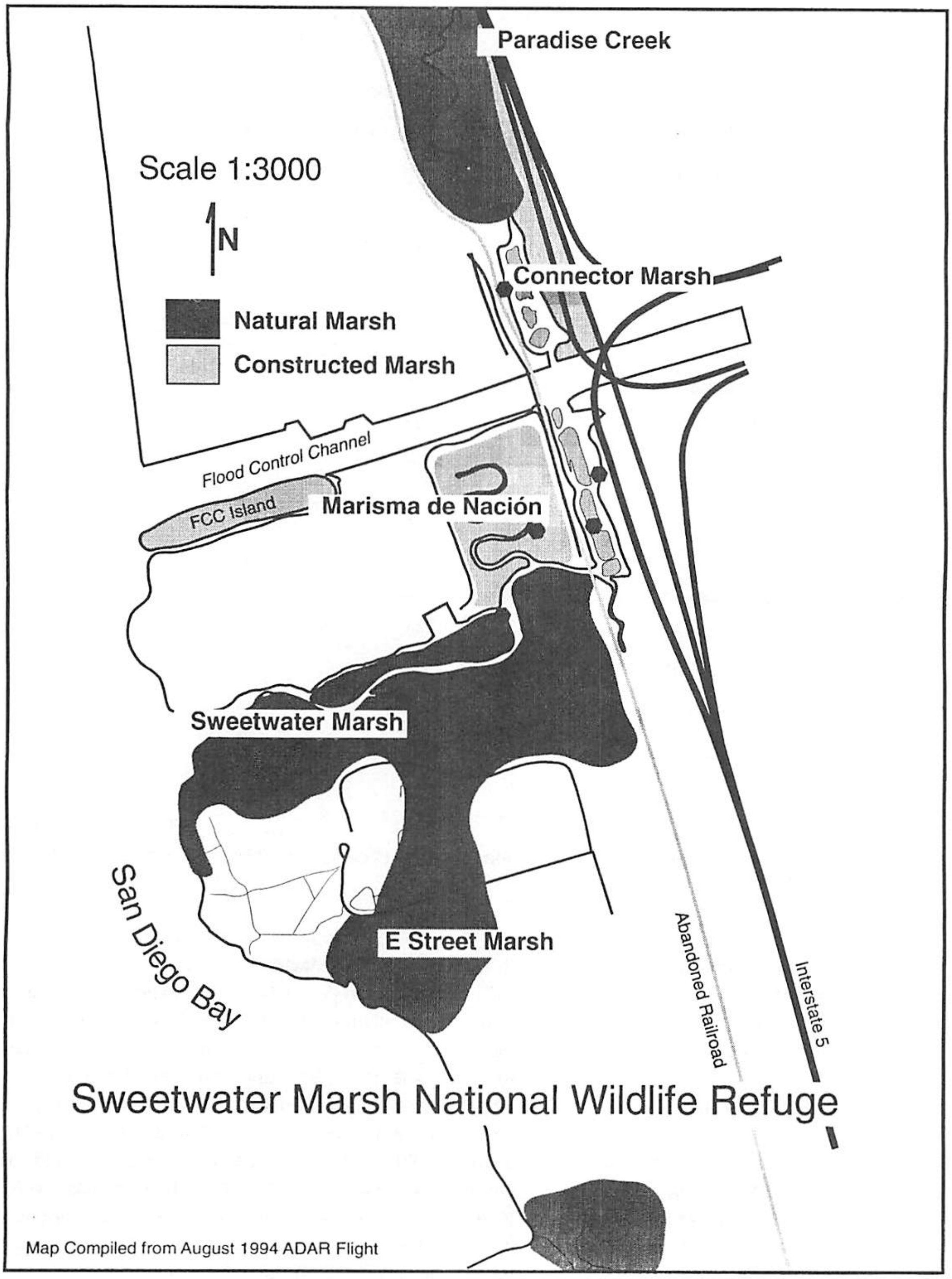


Figure 2.5 Fishing and invertebrate sampling stations.

Table 2.7. Average Percent Cover of Vascular Plant Species Collected Along Nine Transects at Los Peñasquitos Lagoon During September 1990–1993

Species	Percent Cover			
	1990	1991	1992	1993
Salt marsh species				
Cressa truxellensis	5.4	5.3	7.2	4.8
Cuscuta salina	8.9	7.3	17.9	18.5
Distichlis spicata	15.0	7.8	11.7	11.7
Frankenia grandifolia	22.8	22.7	25.7	21.6
Jaumea carnosa	7.8	5.1	4.1	3.3
Monanthochloe littoralis	7.0	3.6	4.1	2.1
Salicornia subterminalis	0.4	0.1	0.7	0.4
Salicornia virginica	52.3	47.0	47.4	47.3
Non-salt marsh species				
Ambrosia psilostachya	0.2	0.4	0.3	0.4
Annual grasses	0.9	1.5	0.6	0.5
Atriplex patula	3.1	2.4	4.4	2.0
Bassia hyssopifolia	0.0	0.1	0.0	0.0
Carpobrotus edulis	1.4	1.7	1.0	2.5
Haplopapus venetus	0.0	0.0	0.0	0.1
Heliotropium curassavicum	0.0	0.1	0.0	0.0
Rumex sp.	0.6	0.5	0.6	0.6
Typha sp.	5.9	2.5	1.5	4.3
Unknown Asteraceae sp.	0.8	0.4	0.4	2.9
Xanthium strumarium	0.0	0.0	0.1	0.7

affinis and *F. parvipinnis* are among its preferred foods. The compliance criteria were met in 1991, when fish species and densities had exceeded 75% of levels in the natural channels for 2 consecutive years. In 1991, 1992, and 1993, all the species that occurred in the natural channel also occurred in the constructed channels (Table 2.9). The total density of these species was slightly higher in the constructed channels. The abundance of exotic species relative to native species remained approximately constant over the monitoring period (1989–1994). The constructed channels of the mitigation projects complied with the criteria for fish as forage for California least terns, and the pattern was sustained through 1994. Sampling continues once a year to obtain a rough, low-cost estimate of fish populations and provide Caltrans with an early warning of any decline in the quality of habitat.

2.3.2 Benthic Invertebrate Assemblage

Benthic invertebrates were sampled periodically at seven stations, six adjacent to fish sampling stations and an additional one in Vener Pond. All stations were visited during January, March, and July in 1989–1993. Cores were collected by pushing a cylindrical "clam gun" (45 cm long, 15 cm in diameter) into the sediment and then sieving the contents through a 1- or 3-mm screen. One set of shallow samples (5 cm deep, 1-mm screen) was collected at each station (three jars, each containing sieved material from three cores) to estimate the abundances of small, shallow-dwelling invertebrates. A second set of deeper cores (20 cm deep, 3-mm screen) was collected to estimate the abundances of large, deep-dwelling invertebrates (mainly bivalves).

In 1992, the invertebrate assemblages (molluscs, crustaceans, and polychaetes) were similar in both types of channels. Each year we collected 9 samples (three stations × three dates) from the natural channels and 12 (four stations × three dates) from the constructed channels. The following rules were established to determine whether the data showed compliance with mitigation criteria: (1) Year-by-year comparison: The numbers and density of taxa present in the natural marsh for a given year were compared with the numbers and densities present in the constructed marsh for the same year. (2) If taxa were present in the natural marsh in previous years but not in the census year, the constructed marsh was not counted. (3) Taxa identified to species were not lumped into the "miscellaneous family" category. Adhering to these rules produced redundancy in some of the data.

Of the 41 native species present in the natural channels, 37 (90%) were also present in the constructed channels (Table 2.10). These species accounted for 1,051 of the individuals found in cores from the natural channels

Table 2.8. Mitigation Criteria to Create a Functional Wetland Complex at Sweetwater Marsh National Wildlife Refuge

1. Tidal channels to provide foraging for the California least tern and light-footed clapper rail.
   - Suitable fish and invertebrate populations shall be present.
   - 75% of the density and diversity of the prey base, compared with that of an existing wetland that does provide habitat, shall be present for 2 years.

2. Low and middle salt marsh for foraging and nesting areas for the light-footed clapper rail.
   - Habitat shall be considered adequate when seven home ranges, nonoverlapping, 2–4 acres, composed of low, middle, and high salt marsh, have been established for 2 years
   - Lower marsh accounts for 15% of the home range and contains 50% cordgrass. Lower marsh in each home range contains one patch of cordgrass with a stem length of 60–80 cm, providing 90–100% cover, that is 90–100 m^2 in size. Cordgrass stands shall be resilient, that is, established for 3 years and with evidence of nitrogen fixation.
   - Middle marsh shall provide 75% cover and contain 75% native species comparable to an existing wetland; middle marsh should include high salt marsh berms.

3. High salt marsh for refugia for the light-footed clapper rail, accounting for at least 15% of each home range.

4. Establishment for salt marsh bird's beak.
   - Patches of salt marsh bird's beak shall be present within the high marsh for 2 years.
   - Patches are considered present when at least five separate patches, each 1 m^2, containing at least 20 individual plants, each separated by at least 10 m, are self-sustaining. Self-sustaining salt marsh bird's beak are stable or increasing in area and number, and present for 3 years

5. Emergent wetlands to provide suitable, functional habitats for the California least tern and light-footed clapper rail, vegetated by 75% of the native species currently present in the Sweetwater River Wetlands Complex.

and 3,450 of the individuals found in cores from the constructed channels. When data were standardized and compared, the number of individuals caught was greater at the constructed channel sites than at the natural channel sites (246%). In the second year, 1993, 85% of the native species present in the natural channels were also present in the constructed channels, and the number of individuals in the constructed channels exceeded those in the natural channels (151%). Thus, the invertebrate communities met the assessment criteria in 2 consecutive years, 1992 and 1993, as well as in 1994 (Table 2.10).

Although crabs are important prey of the light-footed clapper rail (Jorgensen, 1975), coring methods used before 1993 were inadequate for assessing their abundances. In 1993, attempts to trap crabs with minnow traps were moderately successful, although many traps were swept away by currents or were stolen, and once the traps were empty after a very cold night. Overall, eight times as many *Hemigrapsus oregonensis* were caught in the constructed marsh as in the natural sites. Earlier, Rutherford (1989), who used litterbag traps within cordgrass-dominated areas of Connector Marsh, found similar results. Her constructed marsh sites had 2.5 times as many crabs as natural marsh sites in Paradise Creek. The 1993 and 1988 data sets suggest that an abundance of crabs is available to the light-footed clapper rail.

2.3.3 Long-Term Concerns

The speed with which the fish and invertebrate assemblages met the mitigation requirements (relative to salt-marsh criteria, cf. section 5.2) indicates that creation of channel habitat is quite straightforward. Our findings are similar to those of Havers et al. (1995), who found similar fish and shellfish abundances in a constructed tidal marsh and two nearby reference marshes in Virginia. Although they found minor differences due to water salinity, evidence was scant that the constructed marsh was less able to attract native species.

Two remaining concerns go beyond compliance with mitigation requirements. The first is that presence of species does not necessarily indicate functional support of those species. Documentation of fish-support functions would require showing that the channels provide food for fishes, refuges from predators, and places for egg-laying (e.g., macroalgae for attachment of topsmelt eggs). The food-production and refuge functions may require the presence of submerged vegetation or heterogeneous microtopography. Such indicators of function have not yet been incorporated into mitigation requirements.

A second concern with constructed channels is that the disturbances associated with dredging will make channels susceptible to invasion by exotic invertebrates and that these in turn might prevent the normal development of the natural invertebrate assemblages. *Musculista*

Table 2.9. Comparison of Fishes Caught in the Natural and Constructed Channels at Sweetwater Marsh National Wildlife Refuge

Year	Number in Natural Channel	Number in Constructed Channel	%	Mitigation Criterion Met?
1989				
Species	13	8	62	No
Individuals	3.410	2.560	75	Yes
1990				
Species	14	13	81	Yes
Individuals	5.845	10.426	178	Yes
1991				
Species	11	11	100	Yes
Individuals	1.724	4.896	284	Yes
1992				
Species	6	6	100	Yes
Individuals	0.665	0.805	121	Yes
1993				
Species	5	5	100	Yes
Individuals	1.580	4.540	287	Yes
1994				
Species	5	4	80	Yes
Individuals	2.330	2.010	86	Yes

Note: Species numbers are native species found in the natural marsh and also in the constructed channels; "individuals" refers to densities per square meter, averaged for one to four samples per year. Sampling effort was greater for the constructed channel and differed among years: quarterly in 1989–1990, twice in 1991 and in 1992, and once in 1993 and in 1994. Mitigation criteria were that at least 75% of both species and individuals be present for 2 years.

senhousia, the exotic Japanese mussel, is common in natural channels, but it is six times more abundant in the constructed channels. The abundance of this exotic species relative to native species varied over the monitoring period (1989–1993); however, it never exceeded 15% of the total number of invertebrates in the constructed channels (Table 2.11). Another exotic species, *Palaemon macrodactylus*, the oriental shrimp, was caught in the fish seines and was also common in the constructed channels. These exotic species continue to be monitored to detect changes in their numbers or distribution.

In general, we recommend that functional support be assessed more directly than by measuring the presence of species and that long-term assessment of fish and invertebrate habitats be done to ensure short-term success is maintained. Activities that could aid in assessment include determining the feeding habitats of fishes (e.g., gut content analysis), observing predation (e.g., foraging success rates of egrets and herons), and analyzing in detail the microtopography (e.g., tidal creek networks) and structural complexity of submerged vascular-plant assemblages (*Zostera marina, Ruppia* sp.), including macroalgae.

2.4 REESTABLISHING AN ENDANGERED PLANT IN SAN DIEGO BAY: Salt Marsh Bird's Beak

Lorraine Parsons and Joy Zedler

Development of Southern California's coastline has eliminated habitat for a state and federally listed endangered plant, salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus*). Fifty years ago, the species

Table 2.10. Comparison of the Number of Invertebrate Assemblages in Natural and Constructed Channels at Sweetwater Marsh National Wildlife Refuge

	Number in			
Year	Natural Channel	Constructed Channel	%	Criterion Met?
1989				
Species	20	12	60	No
Individuals	52.2	28.8	43	No
1990				
Species	36	25	69	No
Individuals	79.1	119.0	150	Yes
1991				
Species	24	17	71	No
Individuals	85.0	52.0	61	No
1992				
Species	41	37	90	Yes
Individuals	117.0	288.0	246	Yes
1993				
Species	34	29	85	Yes
Individuals	164.0	247.0	151	Yes
1994				
Species	8	6	75	Yes
Individuals	11.0	76.5	695	Yes

Note: Numbers are the total number of native species and the density of individuals (number per 530 cm^2) in the natural channels and the number and density of these species in the constructed channels. The percentages are the constructed channel number expressed as a percentage of the natural channel number. The final column indicates whether the functional equivalency criterion was met. See text for decision rules used to compile this table. Sampling efforts in the natural and constructed channels were different each year. The severe decline in individuals and abundance in 1994 is attributed to the use of alcohol instead of formalin as a preservative. Rapid decomposition prevented identification to the species level.

was found in 18 Southern California coastal marshes and was characterized as a "frequent" inhabitant of those in San Diego County (Purer 1942). Now, only Tijuana Estuary sustains its natural population. Plants at Sweetwater Marsh were introduced to fulfill a Caltrans mitigation requirement (U.S. Fish and Wildlife Service 1988) for reestablishment of a self-sustaining population of bird's beak.

In 1990, PERL began efforts to reestablish the species. Because this species had not been observed at Sweetwater Marsh since 1987, seeds were taken from colonies at Tijuana Estuary and sown at sites in the marsh from 1990 through 1992. Bird's beak was initially sown in a remnant high-marsh area constructed as part of the mitigation project, but germination and seed set of the sown colony were low in 1990. As a result, Caltrans received approval to move the reestablishment project to a "natural" high marsh south of the constructed marsh. Sowings were conducted in four "colonies" in 1991; three were reseeded in 1992 (B. Fink, personal communication). During 1990 and 1991, germination and growth of bird's beak sown at Sweetwater Marsh appeared comparable to that of the seed donor population at Tijuana Estuary (B. Fink, personal communication). Seed capsule set, however, was minimal at several of the Sweetwater Marsh colonies, especially when compared with rates of the seed donor colonies at Tijuana Estuary (B. Fink, personal communication; Fink and Zedler 1991).

In 1991, seed capsule set reached 80% in some colonies at Sweetwater Marsh, but other colonies had rates of 20% or less (Fink and Zedler 1991). Meanwhile, the

Table 2.11. Abundance of Exotic Species of Fish and Benthic Invertebrates in Natural and Constructed Channels at Sweetwater Marsh National Wildlife Refuge During 1989–1993

Organism	Percent in Natural Channel	Percent in Constructed Channel
Exotic fishes caught in seines		
1989	1	3
1990	0	1
1991	3	2
1992	0	1
1993	5	1
Exotic fishes caught in traps		
1991	9	15
1992	12	12
Exotic invertebrates		
1989	0	0
1990	8	5
1991	17	9
1992	3	4
1993	3	14

Note: Data are percentages of totals collected.

seed donor colonies at Tijuana Estuary had seed capsule set rates as high as 90% (Fink and Zedler 1991). In 1992, average seed capsule set of plants at Sweetwater Marsh was 19% (± 2%), with values ranging from 0% to 75%. Because the number of seeds produced represents the most relevant indicator of long-term population change for annual plants (Ricklefs 1990), low seed set of *C. maritimus* spp. *maritimus* could jeopardize efforts to establish a self-sustaining population.

Concerned about the impact of poor seed-capsule set on the restoration, Caltrans funded a study of factors that affect reproductive potential of bird's beak at Sweetwater Marsh, including effects of herbivory, pollination, environmental factors, and intraspecific competition on mortality, growth, and production of flowers and seed capsules. Pollination and the availability of nitrogen affected reproductive potential the most, and additional research on these two factors was funded by the U.S. Fish and Wildlife Service.

2.4.1 Ecology of Bird's Beak

Salt marsh bird's beak is an annual species that thrives in the high-marsh areas of salt marshes ranging from Morro Bay to Baja California. Bird's beak germinates from February through March and flowers from May through August or early September, becoming senescent shortly thereafter. No study has specifically investigated the conditions necessary for breaking seed dormancy in bird's beak, but germination appears to be triggered by winter and spring rains. Opening of dense canopies of vegetation encourages expansion of bird's beak (Vanderwier and Newman 1984, Fink and Zedler 1991), often creating a mosaic of dense plant clusters or patches within colonies.

Bird's beak is a facultative hemiparasite, which means that the plant can derive at least some sustenance from other plants (host species). Bird's beak and other members of the genus *Cordylanthus* tap into host plants or even other bird's beak plants through small, ephemeral root connections called haustoria (Chuang and Heckard 1971, Vanderwier and Newman 1984). Plants may receive water and nutrients from host species, but no study has qualitatively determined just what type of or how much sustenance is received. Fink and Zedler (1990b) found that plants grown with hosts produced significantly more biomass (six to eight times more) and had longer growing seasons (more than 4 weeks) than plants grown without hosts and that bird's beak appeared to grow better with some host species than with others (*Distichlis spicata* > *Monanthochloe littoralis* > *Salicornia virginica*; Figs. 2.6 and 2.7) However, Vanderwier and Newman (1984) found "no observable increase in vigor" in plants grown with or without hosts. Both of these studies were done in a laboratory.

Distribution. Bird's beak grows best within a narrow band of elevation, from approximately 6–7 ft (1.8–2.1 m) mean lower low water (MLLW) (U.S. Fish and Wildlife Service 1984, Fink and Zedler 1990b). At Tijuana Estuary, the mean elevation of patch perimeters (116 plots were surveyed in 1994) was 7–12 ft (2.2 m) MLLW (0.04 SE). At Tijuana Estuary, bird's beak occurred in areas with microtopographic relief (small mounds 10–20 cm high), and plants were found growing both on top of mounds and between them (Fink and Zedler 1990b). At Sweetwater Marsh, almost all plants grew between such mounds, rather than on top. Fink and Zedler (1990b) postulated that microtopography might be important in reducing dispersal of the seed crop while allowing rainfall to puddle and dilute soil salinity. Mortality of bird's beak has been positively correlated with higher elevations, drier soils, and lower levels of clay in the soil (Parsons and Zedler 1993).

At Sweetwater Marsh, *Monanthochloe littoralis* accounted for more than 80% of the host plant cover in 1992. A positive relationship might be expected between cover of preferred hosts and occurrence of bird's beak. However, in the field, aboveground biomass and density of bird's beak correlated negatively with cover of host species (Parsons and Zedler 1993). At higher densities, host plant species may compete with bird's beak for light, moisture, or nutrients. Relationships with host plants may promote seedling survival and vegetative growth by allowing bird's beak to obtain water from more deeply rooted species. Soil moisture is scarce in spring, because tides rarely exceed 6.5 ft (2.0 m) (Fink and Zedler 1990a). At Tijuana Estuary, only 8 of 16 surveyed plots were

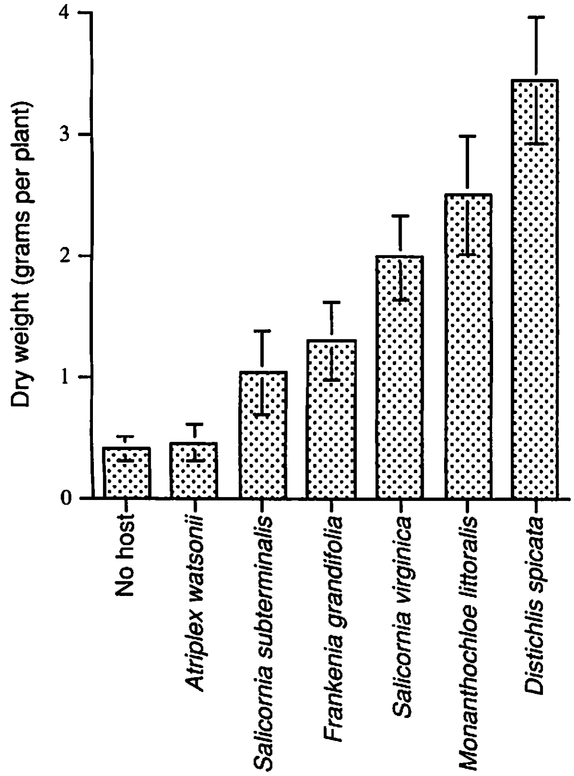


Figure 2.6 Aboveground growth of salt marsh bird's beak (*Cordylanthus maritimus* spp. *maritimus*) alone and with six host plant species. Bars are ± 1 SE. (From Fink and Zedler 1990).

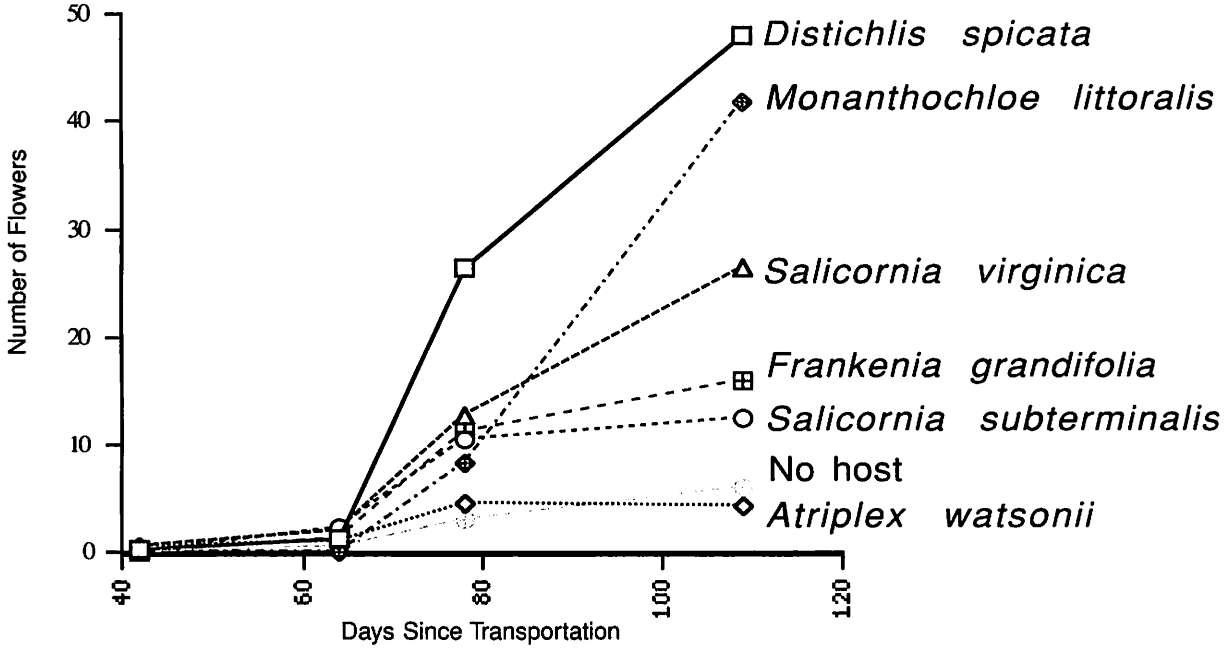


Figure 2.7 Differences in flower production by salt marsh bird's beak (*Cordylanthus maritimus* spp. *maritimus*) when grown with various host species and when grown alone. Redrawn from Fink and Zedler 1990.

less than 6.5 ft (2.0 m) MLLW. Despite the inherent fragility of hemiparasitic relationships, bird's beak shows plasticity in the types of species parasitized. This plasticity led Vanderwier and Newman (1984) to conclude that "salt marsh bird's beak distribution is a function of strong habitat preference rather than of host preference."

Bird's beak is patchy, occurring sometimes in high-density groups and often as isolated individual plants. Although intraspecific competition would not seem to be a problem for an endangered species, sowings in 1992 did create patches in which seedling densities reached 220 seedlings/decimeter^2. As with many other plant species, increasing density did not increase mortality, but it did lower the biomass of individual plants (Parsons and Zedler 1993). Reduced biomass can then lower flower number and seed capsule set (Parsons and Zedler 1993).

Requirements for growth. Soil substrate may be crucial for seedling recruitment and reproductive potential and success as well. In our study, plant mortality was lower in areas with more clay, undoubtedly because of the higher water-holding capacity of fine-textured soils (Mengel and Kirkby 1982). Aboveground biomass of plants also correlated strongly with levels of inorganic nitrogen in soils.

Treatments with nitrogen fertilization dramatically increased plant biomass (>100%), foliar nitrogen content (21%), and seed size (18%) and appeared to increase flower number (200%) relative to that of unfertilized plants. When fertilized with nitrogen, plants appeared to invest these resources differentially by increasing the number of branches, inflorescences, and flowers (Parsons and Zedler 1994). Ultimately, increasing the number of inflorescences and flowers increases the plant's reproductive potential, because pollinators forage preferentially on larger plants and on plants with more flowers (Parsons 1994).

The salinity of the soil did not appear to affect performance of bird's beak (a halophyte). We do not know whether plants were subject to stress because of salinity; however, salinities never exceeded 33 ppt in saturated soil pastes prepared from marsh soil (Parsons 1994). Rainfall was high during the 2 study years, thereby decreasing salinities in relation to dry or average years (Parsons 1994). In a laboratory experiment, bird's beak plants treated with water with a salinity of 36 ppt had only a slightly lower dry weight than those treated with fresh water (3 ppt) (Fink and Zedler 1990b). However, the method may have also minimized soil salinities. Soil salinity was analyzed by using soil pastes (Richards 1954) because some samples were very dry, and salinities may have been artificially dampened by resaturation of oven-dried soils.

Insect herbivory. A primary insect herbivore of salt marsh bird's beak at Sweetwater Marsh was the larval stage of *Lipographis fenestrella* (Pyralidae) or salt marsh snout moth. This same species has been observed feeding on bird's beak in Tijuana Estuary and Point Mugu (Nagano and Hogue 1982, Parsons 1994). While damage to plants often appeared extensive, at Sweetwater Marsh, granivory (eating of seeds by larvae) did not adversely affect the percentage of flowers producing seed capsules, the percentage of flowers producing mature or undamaged seed capsules, or the number of seeds produced per capsule (Parsons 1994, Parsons and Zedler 1994). In many cases, even seed capsules showing signs of granivory (i.e., bored holes in the seed coat) still retained undamaged seeds.

Granivory was extremely variable at Sweetwater Marsh, devastating some patches and minimally affecting others. The reason for this selective pattern is not known, but granivory showed a highly significant relationship with flowering time of patches. Early flowering patches (late April to early May) had more seed predation than late flowering ones (late May to early June) (Parsons 1994). The phenology of these early flowering plants may coincide more closely with the oviposition cycle of the egg-laying micropyralid moth than that of the later flowering plants.

Pollination. Insects also play a positive role in population dynamics of bird's beak. As an insect-pollinated annual, bird's beak relies primarily on insects for reproduction. The main pollinators of bird's beak appear to be bees from the genera *Bombus* (bumblebees), *Halictus*, *Lasioglossum*, *Anthidium*, and *Melissodes* (U.S. Fish and Wildlife Service 1984, Lincoln 1985, Parsons 1994). Low rate of seed capsule set at some colonies, combined with observations of few pollinators, led Fink and Zedler (1991) to postulate that a lack of pollinators could be limiting seed set at Sweetwater Marsh. Development of adjacent upland areas may have eliminated nesting or alternative foraging habitat for insect pollinators. In both 1992 and 1993, treatments to supplement pollination (hand pollination) significantly increased seed capsule set. Hand-pollinated plants set 89% more and 52% more seed capsules than naturally pollinated plants in 1992 and 1993, respectively (Parsons 1994).

Sheer abundance of pollinators did not appear to be as important to pollination success at Sweetwater Marsh as the relative abundance of pollinating species. Plants in patches visited by either *Bombus* and halictines or *Melissodes* and halictines had higher rates of seed capsule set than plants in patches visited by halictines alone (Parsons 1994). *Anthidium edwardsii*, characterized as the most important pollinator of bird's beak at Point Mugu and Tijuana Estuary, was not observed at Sweetwater Marsh in either 1992 or 1993 (Parsons 1994). No historical information is available to indicate whether *Anthidium* was present at Sweetwater Marsh when the historical population of bird's beak existed.

Results from the hand-pollination treatments sug-

gest that, generally, low levels of pollination may have had the most impact on reproductive success of bird's beak at Sweetwater Marsh. Proximity of plants may have affected foraging behavior more than resource abundance. Pollinator visits showed both a strong block (patch) and cluster (grouped vs. isolated patches) effect but did not correlate with the density of inflorescence (Parsons and Zedler 1993). Pollinators may preferentially visit patches on the basis of proximity to nests or clustering of patches rather than on the basis of the amount of flowers within a patch, or pollinators may simply encounter clustered patches more often than isolated ones, regardless of the numbers of flowers present. Except for *Bombus,* most of the bees pollinating bird's beak are thought to nest near their foraging sites. Halictine and *Melissodes* bees build nests below ground, whereas *Anthidium* and *Bombus* build nests in preexisting cavities, either above or below ground (Laberge 1956, Stephen et al. 1969). The density of inflorescence did affect the number of plants visited per patch (Parsons and Zedler 1993). Pollinators preferred medium- and large-sized plants to small ones, probably because biomass correlated positively with flower number (Parsons and Zedler 1993).

2.4.2 Monitoring for Mitigation Requirements

In 1992, five sites, each containing five small patches of bird's beak, were selected for annual monitoring at Sweetwater Marsh. These groups of patches met the spatial requirements outlined in the mitigation criteria (at least 10 m apart). In order to determine the most accurate and time-efficient method of monitoring bird's beak, three sampling methods were used in July 1993 to assess bird's beak numbers: quadrat density, distance (first to second plant), and point-quarter (Parsons and Zedler 1993). Because the distribution of bird's beak is patchy, methods were selected on the basis of their sensitivity to nonrandom distributions, and corrections for clumped distributions were used if available. All sampling methods, however, do assume random distribution of targeted plant species, which complicates plant sampling. As a control, all plants within the area delimited by the outer perimeter of the five designated patches were counted three times, and the three counts averaged. These numbers were then compared with estimates obtained by using the other three sampling methods. When compared with total count averages, numbers of bird's beak plants were underestimated with all three sampling methods, sometimes by a factor of 100 or more (Parsons and Zedler 1993). Therefore, an average of three total counts of each group of patches was used for monitoring the number of plants present.

A 3-year period with at least 100 plants is required under the mitigation agreement (Table 2.8), and the censusing of bird's beak began in 1993. We estimated population size and the total number of seeds produced by plants present in both 1992 and 1993. Unfortunately, we could not determine germination rates, because seedlings may have come from seeds sown by Fink in 1990, 1991, and 1992, as well as from seeds produced by the new population of plants. Seeds are known to remain viable for more than a year.

The five sites all exceeded the minimal requirement of 100 plants in both 1993 and 1994, with more than 1,000 plants in the monitoring area in 1993 and more than 2,000 in 1994. Estimates of the numbers of plants in the entire Sweetwater Marsh were more than 5,000 in 1993 and more than 14,000 in 1994. These estimates include plants at the original sowing site (southernmost Connector Marsh island) and two sites in the northern Connector Marsh islands (cf. Fig. 2.5) that were not planted. Although plants at the last two sites were not numerous (<100 in 1993 and about 500 in 1994), their appearance indicates potential for spread into constructed marsh habitats, at least where host plants and good soils occur.

The main problem in the constructed marsh is that the suitable elevations tend to be unvegetated, highly saline flats. Average salinity in these bare spots was more than 130 ppt in August 1994, compared with 46 ppt in vegetated high marsh. Thus, while reestablishment of bird's beak in remnant high marshes appears feasible, developing a self-sustaining population in constructed marshes, especially those on sandy dredge spoils, is less likely. Attempts to do so will begin after 1995, when we anticipate that the mitigation criteria will be met and seeds can be collected and sown at Marisma de Nación.

2.4.3 Implications for Monitoring and Reestablishment of Bird's Beak

Recently, Schemske and colleagues (1994) argued that the "research recommendations for federally listed plants are, in general, not sufficient to identify the biological information required to develop sound recovery guidelines." They advocate a three-step approach to studying rare and endangered plants, determining (1) if the population size is increasing, decreasing, or stable; (2) which life stages have the greatest effect on population growth and species persistence; and (3) the biological causes of variation in those stages that have a major demographic impact.

Implementing this approach is difficult. In the case of *C. maritimus* spp. *maritimus*, we could not determine the most critical life stage. Because of differences between seed capsule production at Sweetwater Marsh and Tijuana Estuary, the stage that most likely regulates numbers of bird's beak at Sweetwater Marsh was seed set. Our rough estimates of seed production (more than 700,000 in 1993 and 1994) also suggest that germination plays a crucial role in regulating population size of bird's beak at Sweetwater Marsh (Parsons and Zedler 1994).

Data to determine the population size needed for long-term persistence are unavailable. Annuals with persistent seed banks, such as bird's beak, probably experience annual fluctuations. Long-term monitoring would be needed to determine if the population can survive years of severe environmental stress, and few research or management projects have the necessary budget or life span.

Counting plants is not enough. Managers need to be able to predict when population declines indicate something more than interannual variability and to correct limiting factors before the decline becomes irreversible. Ensuring adequate pollination, seed production, and germination is critical to maintaining bird's beak at Sweetwater Marsh. Conservation of the species requires that the essential bee species have suitable habitat near the marsh, that natural processes that supply nutrients to the high marsh be determined and restored, that the natural salinity regime be sustained, and that natural disturbances that open small-scale patches in the high-marsh canopy be present.

2.4.4 Recommendations for Introducing Salt Marsh Bird's Beak

Future efforts to introduce salt marsh bird's beak to sites where it once existed or to constructed wetlands may benefit from the following recommendations:

- Seed-sowing densities should be controlled to avoid intraspecific competition. Sowings should aim for intermediate densities (30–50 seedlings/decimeter^2), because such densities may maximize biomass without losing the productivity associated with higher densities. High densities of inflorescences within patches do not appear necessary for attraction of pollinators.
- Seeds should be sown to create clusters of patches rather than isolated ones.
- Cover and type of potential host plant species should be considered when creating patches. Preferred hosts include *Distichlis spicata*, *Monanthochloe littoralis*, and *Salicornia virginica*.
- Thinning may be needed to create small openings in the high-marsh vegetation and encourage expansion of bird's beak. Cover of potential host plants should not be too dense, because it may compete with bird's beak for light or other resources.
- Seed should be sown at elevations between 6–7 ft (1.8–2.1 m) MLLW and where soil is high in clay and low in sand.
- A link between the plant's high-marsh habitat and upland areas needs to be present, because upland areas may be used for nesting and foraging by important pollinators such as *Bombus*. Further research is needed to clarify the nesting requirements of pollinators in salt marshes.
- Because of the sensitivity of most vegetation sampling methods to patchiness in plant distribution, total counts should be used to assess numbers of bird's beak within groups of patches selected for monitoring. Counts should be done in late June to early July, when most plants have completed vegetative growth but before any senescence occurs.

2.5 LESSONS FROM CASE STUDIES

Dana Johnson

In this section, brief case studies of nine restoration sites ranging from California to British Columbia are presented. Management goals for the eight sites being undertaken under mitigation requirements appear in Table 2.12. As the review will illustrate, unanticipated problems made these goals difficult to achieve in every case.

Individual lessons are drawn from each study, and generalizations can be made for future restoration efforts. It should be evident that it is difficult to duplicate the most basic determinants of biological systems—namely, the hydrology and substrates of natural wetlands—when managers have little choice of location or materials. It is also evident that most restoration efforts are inadequately documented, and that detailed scientific studies have not generally been made. Further, although we have some idea of the problems that arise during restoration, their causes and biological impacts are poorly known.

The great potential for restoration activities is suggested by long-term observations of constructed or modified wetlands that show use by native species. A reasonable conclusion is that we can create habitat, but we cannot always predict what it will be like or what species will benefit. There is obviously considerable room for improving the predictability of wetlands restoration and creation.

Table 2.12 Criteria for Compliance with Mitigation Requirements

Warm Springs Marsh	• Provide flood protection • Improve water quality • Provide for public enjoyment • Enhanced habitat for wildlife, specifically pickleweed (*Salicornia virginica*) habitat for the salt marsh harvest mouse (*Reithrodontomys raviventris*)
Bair Island Ecological Reserve	• Return tidal flow to the site • Enhance the habitat for wildlife use • Manage the reserve as a least tern nesting area
Hayward Regional Shoreline Marsh	• Create cordgrass (*Spartina foliosa*) habitat • Create open water habitat for waterbird use • Convert most of the site to a tidal salt marsh
Muzzi Marsh	• Return tidal flow to the site • Establish a self-sustaining habitat for waterfowl and shorebirds
Humboldt Bay Mitigation Marsh	• Replace lost shoreline, subtidal, and intertidal habitats
Salmon River Estuary	• Return tidal flow to the site • Return the estuarine system to conditions existing prior to diking and agricultural use
Gog-Li-Hi-Te Wetland System	• 50% use by juvenile salmon • 50% use by waterfowl, shorebirds, raptors, and small mammals • Transplant *Carex lyngbyei* • Monitor for five years
Pacific Terminals Saltmarsh	• Construct a saltmarsh habitat to meet the no-net-loss policy of the Canadian government

Los Peñasquitos Lagoon

Lessons: 1. An adaptive management program is necessary to track water quality so the ocean inlet can be opened and fish kills avoided.
2. Frequent small-scale interventions (bulldozing the mouth annually) may delay or eliminate the need for massive dredging.

Los Peñasquitos is a relatively small estuary (142 ha) in northern San Diego County that is frequently closed to tidal flushing. An inlet-dredging program has been developed to alleviate this problem.

The lagoon includes 86 ha of salt marshes and 13 ha of tidal channels. It provides habitat for many species of estuarine plants, invertebrates, fish, and birds. Several sensitive species are present in the lagoon, including moderate populations of the Coulter goldfields (*Lasthenia glabrata coulteri*) and the endangered Belding's savannah sparrow.

The main environmental problem at the lagoon is the frequent closing of the mouth by beach sand and cobble. These closings cause widespread damage in the lagoon; channel waters often become stagnant, leading to extensive killing of fish and invertebrates. The subsequent accumulation of inflows can cause the salt marsh to become flooded, making it unsuitable for breeding birds and causing the deaths of many plants and the elimination of some species.

In 1985, an enhancement plan and program was prepared by the Los Peñasquitos Lagoon Foundation and the California Coastal Conservancy. This plan focused on solving the problem of the frequent mouth closings. It was estimated that the lagoon had been a tidal estuary in the 1920s but that construction of a railroad and a road through the estuary in the early 1900s had altered the hydrology to such an extent that ebb flows could no longer maintain a mouth channel. An extensive dredging program designed to increase the tidal prism could have been undertaken, but this was considered too destructive to the existing environment. Instead a cautious and experimental approach was used to restore tidal flow. By 1992, a strategy had been developed that resulted in the mouth of the lagoon being open 11 of 12 months. This result appears to be due to a better appreciation of the mechanisms of sedimentation around the mouth.

The beaches around Los Peñasquitos Lagoon are composed of sand overlying cobble. Winter storm waves remove the sand from the beaches and deposit it in the subtidal area, thereby exposing the cobbles. The smaller waves of spring and summer return the sand to the beach and cover up the cobbles (Kuhn and Shepard 1984). The mouth of the lagoon is part of this dynamic system. During the winter, cobbles are pushed into the mouth channel, and during spring and summer, sand tends to cover them. In the 1980s, the cobbles formed a sill in the mouth at approximately mean sea level that prevented some inflows, slowed ebbing flows, and formed a paving on the bottom of the channel that impeded erosion, thereby preventing the ebb flows from creating a deep, long-lasting mouth channel. The mouth generally closed soon after the winter rains when sand began to be deposited again.

Los Peñasquitos Lagoon Foundation's approach in the 1990s has been to remove the cobble sill at the mouth. During the springs of 1991–1994, cobbles in the mouth channel were removed by using a big-wheel front loader and a self-loading scraper, and sand and cobbles were deposited on the beach approximately 200 m south of the mouth.The work usually took 5 days. The mouth of the lagoon had been open approximately 25% of each year during the 1980s, but it was open 48% of 1991, 95% of 1992, 91% of 1993 (Boland 1993b), and at least through September 1994. Better tidal flushing has led to several habitat improvements, particularly in water quality.

Warm Springs

Lessons: 1. The nature and direction of hydrologic changes cannot be predicted with certainty.
2. Monitoring is essential in wetland management.

The project to restore Warm Springs Marsh, located in the southern part of San Francisco Bay (Fig. 2.8), was a cooperative mitigation effort of governmental agencies, private developers, consultants, and an independent environmental group. Originally considered high-marsh plain, the 100 ha site chosen for restoration was located within an established salt marsh. For 40 years the area was diked for agricultural use and salt harvesting. Pumping of groundwater, which caused compaction and oxidation of the marsh soil, combined with discharges from industrial and sewage treatment plants, created a highly degraded condition (Williams and Morrison 1988).

The restoration plan was developed to facilitate the evolution of a natural marsh with as little human intervention and maintenance as possible. The broad objective was to create a system capable of providing the functions characteristic of wetlands, that is, flood protection, improved water quality, public enjoyment, and habitat for wildlife. The design specifically focused on enhancing pickleweed habitat, home to the endangered salt marsh harvest mouse (*Reithrodontomys raviventris*) (Williams and Morrison 1988). In 1986, excavation created a convoluted shoreline and a large embayment separated by peninsulas into six lagoon cells (Abbe et al. 1991), and northern and southern levees were breached, joining the site to the existing marshes.

Monitoring was not a condition of the original permit. In response to an obvious need for documentation, a detailed monitoring program that spanned 3 years was subsequently funded by the San Francisco Foundation (Williams and Morrison 1988). Conditions existing before breaching and characteristics of the restoration were documented at least twice. Variables examined included elevations and cross-sections, tidal patterns, salinity, channel erosion, sedimentation, tidal circulation, and use by plants and animals.

The substantial increase in the volume of water entering the southern part of the lagoon caused high rates of sedimentation. Surveys of cross-sections of southern and northern lagoons showed rates of sedimentation between 1.1 m/year and 0.2 m/year, respectively (Abbe et al. 1991).

Pickleweed (*Salicornia virginica*), fat hen (*Atriplex patula*), and cordgrass (*Spartina foliosa*) seeded themselves. Pickleweed cover has increased with time in the upper elevations; cordgrass has replaced pickleweed in the lower elevations (Abbe et al. 1991). In 1990, 4 years after the initial breaching, it was apparent that vegetative colonization and distribution were responding to the changing physical conditions of the site. In 1988, Williams and Morrison observed that "over 70 species of birds [use] the restoration site, including flocks of white pelicans feeding, and the peregrine falcon (designated by the federal government as an endangered species)." Three species of fish use the constructed channels for breeding, and several others have been observed in the area. Harbor seals have also been seen in the embayment.

As the site evolves with the changing flow patterns, monitoring has shown unexpected "feedback responses not [anticipated] in the original design" (Abbe et al. 1991). In 1991, the state of the ecosystem was still considered dynamic and its future difficult to predict. "Continued monitoring of the restoration [is] critical in assessing the relative performance of the site as it was designed" (Williams and Morrison 1988).

Bair Island Ecological Reserve

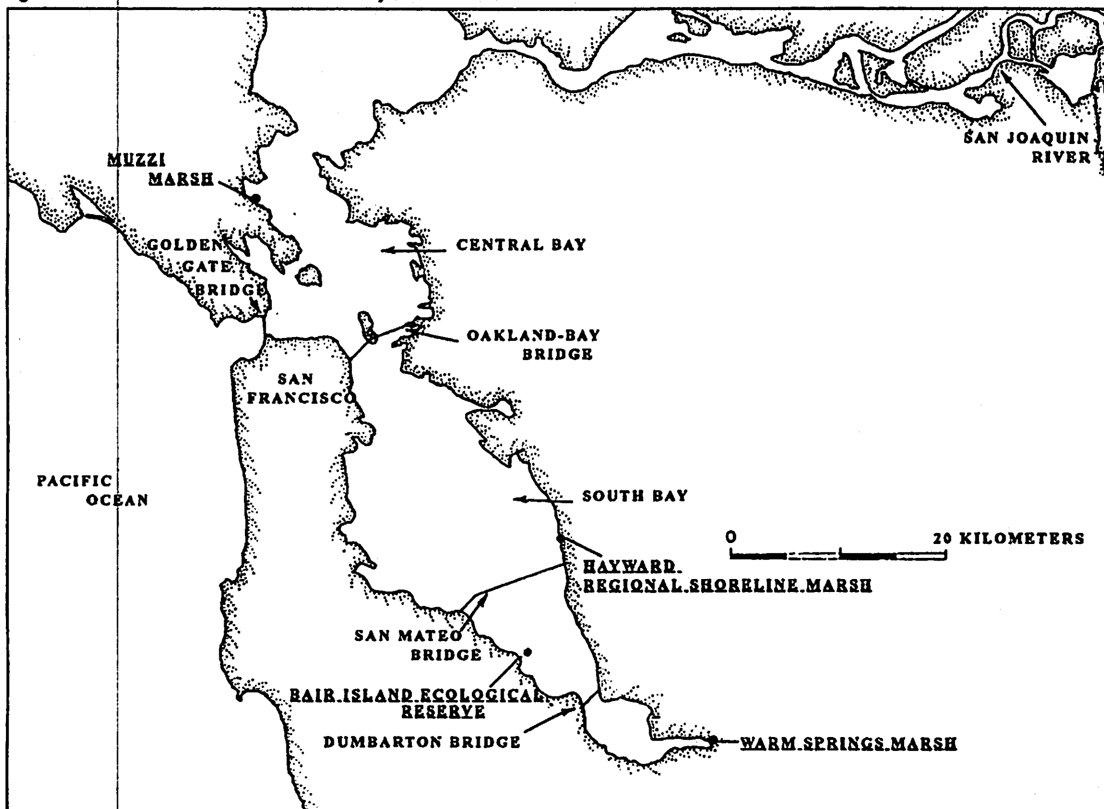
Lesson: 1. Management goals may be slow to evolve.

One hundred years ago, Bair Island was a salt marsh in the southwestern part of San Francisco Bay. Parts were then diked for agriculture until 1948, at which time most of the remaining island was diked for salt evaporation ponds. In 1973, the land was acquired by Mobile Oil Estates. The California State Lands Commission obtained 324 ha of the island as part of the transaction. The Department of Fish and Game "determined that the site was an important habitat for wildlife and accepted a lease with management responsibilities for the site" (Josselyn et al. 1991). In 1974, an agreement was made by Mobile Oil Estates, the Department of Fish and Game, and the State Lands Commission to conduct a comprehensive 3-year study of the environmentally sensitive areas within the site, including "recommendations on the future use and/or preservation of such areas" (Josselyn et al. 1991). In 1976, additional acreage was donated by the owner as mitigation for the development of former marshlands.

Environmental reports indicated numerous existing and potential wildlife values. Restoring tidal flow to the site and recreating a salt marsh habitat "further enhanced the site's value" (Josselyn et al. 1991). Between 1973 and 1976, the Department of Fish and Game discussed options for restoring tidal action to parts of Bair Island. In 1976, an unplanned breach in an outer levee flooded a large part of pond B1. As a consequence of the winter storms and prolonged argument, baseline data on the physical and hydrologic characteristics of the area were never collected. Likewise, an attempt by the California Conservation Corps to transplant cordgrass to the flooded area was undocumented.

Despite opposition by developers, the U.S. Army Corps of Engineers issued a permit to manage the site as a least tern nesting area. Active management by the San Francisco Bay Bird Observatory began in 1980. In the next 3 years, different strategies were used to enhance the nesting site. By 1983, it seemed that terns showed no preference for a particular constructed nest, and nests were found in a variety of locations (Josselyn et al. 1991). That same year, concern for the erosion of levees surrounding the breeding colony resulted in numerous conflicting proposed solutions. The plan to construct an internal

Figure 2.8 Locations of San Francisco Bay marshes for case studies.



levee was finally agreed on. Unfortunately, the project has not created conditions as planned (Josselyn et al. 1991). No least terns have used the area since 1988. Deterioration of northern levees threatens to drive Caspian terns from the area as well. Currently, predation by red foxes is a problem. Large parts of heron and egret rookeries were destroyed by red foxes in 1991. In an attempt to attract these birds back to the reserve, the U.S. Fish and Wildlife Service proposed to use "confidence" birds (or decoys) in the winter and spring of 1994–1995.

In 1983, a decision was made to increase tidal flows to the adjacent pond (B2). A channel was cut through the levee separating the two major ponds. Gradually, pickleweed and cordgrass colonized the area. Vegetative cover increased from 73 acres in 1982 to 175 acres in 1988.

To date, four federally or state protected species and three species of special concern have been found within Bair Island. Most of these have been selected as target species for the management goals (Josselyn et al. 1991). The salt marsh harvest mouse has successfully been trapped on and adjacent to the site since 1971. In a survey in the winter of 1993–1994, seven California clapper rails were observed within the Reserve (C. Wilcox, personal communication).

Hayward Regional Shoreline Marsh

Lessons: 1. A large buffer zone is required to facilitate even limited use by wildlife.
2. Slow establishment of vegetative cover may encourage invasive species.

The Hayward Regional Shoreline historically was located on the site of Crystal Salt Pond and an existing marshland. The project to restore the shoreline began as mitigation for the construction of a new bridge across San Francisco Bay. Between 1865 and 1927, and again during World War II, the site, which consisted of more than 81 ha, had been used primarily for salt production.

In 1973, the City of Hayward purchased a large part of the site to create wastewater oxidation ponds. In anticipation of plans to further develop Hayward's 13 km shoreline, several local planning agencies prepared land use reports. Caltrans funded the proposed marsh restoration whose primary habitat objective was to support the establishment of cordgrass. Secondarily, the design incorporated large areas of open water to encourage use by waterbirds. In 1980, extensive movement of earth and dredging of the channel were done. The ultimate goal was to convert most of the site to a tidal salt marsh similar to the one that probably occurred under primitive conditions (Josselyn et al. 1987).

A 15-month study reported in 1981 (Josselyn and Niesen 1981) found no natural establishment of cordgrass. Natural drainage channels had not developed, and large areas of the restoration site contained standing water. In 1983 and 1984, cordgrass culms were planted, but establishment was slow. Erosion in the natural marsh was also gradually reducing the area of vegetated marsh there (Josselyn et al. 1987).

From 1986 to 1987, field studies compared the Hayward Regional Shoreline Marsh with a natural marsh in San Leandro Bay. Data were collected on soil and vegetation conditions, bird use, and fish abundance. Vegetation in the natural marsh was dominated by perennial wetland plants and by a "preponderance of algal mats" (Josselyn et al. 1987). In contrast, the restored marsh had many more annuals (7 of 12 species) and nonnative species (5 of 12), and no algal mat cover. Cordgrass had begun to colonize the open mudflats; however, in the 6 years since work had begun, the cover of cordgrass was less than 10% (Josselyn and Duffield 1987). Pickleweed replaced cordgrass at the upper edges of the marsh. Low vegetation cover was thought to be a result of low soil moisture and organic content.

Poor development of sloughs and channels and high rates of sedimentation in the restored marsh limited its use by juvenile fish. The greatest diversity of fish species was supported within San Leandro Bay. Hayward Regional Shoreline Marsh was primarily used by topsmelt.

The marsh at San Leandro Bay generally had greater water depth at all tide stages and more protected coves than the shoreline marsh. Shorebirds at the reference marsh used the site for roosting and the edges of the marsh for preening and resting. No comparable high tide roosting site was available in the constructed marsh; the artificial islands were used by only a few egrets. Most birds preferred the small available patches of pickleweed during high tide. Public access trails at the Hayward site were considerably closer to wildlife areas than trails in the bay marsh, and no buffer vegetation was present (Josselyn et al. 1987). As a consequence, few birds remained in the Hayward Regional Shoreline Marsh when visitors were present in the park.

In their 1987 report, Josselyn et al. concluded that "project objectives should not focus on any one habitat type . . . as an indication of success." At this writing, the site is heavily invaded by *Spartina alterniflora*, an invasive Atlantic Coast species (M. Josselyn, personal communication, 1994), and no new studies are being done.

Muzzi Marsh

Lesson: Acceleration of restoration may depend on the placement of dredged materials.

For 15 years before its restoration, the site of Muzzi Marsh was diked and used for the deposition of dredge spoils to create sites for commercial development. Reconstruction of the historical elevation indicates that by the mid-1970s the area had subsided 0.2–0.7 m (Gahagan and Bryant et al. 1994). Restoration began as part of a mitigation project for dredging a ferry access channel through the Corte Madera mudflats. At the time, "little was known about the physical or biological processes of marsh restoration" (Williams et al. 1988). However, public support for ecological issues was rapidly growing. In 1976, the Bay Conservation and Development Commission issued a permit to construct Muzzi Marsh. Approximately 573,450 cubic meters of dredged materials were placed in the upper part of the marsh and the bayward part of the site (Gahagan and Bryant et al. 1994), and levees were breached to return tidal flows to 53 ha of former wetlands. This marsh was one of the first such restoration projects in California, and its evolution has been monitored with great interest and in some detail over the past 14 years.

The area covered by pickleweed increased quickly. Plants spread from the occasional clump in 1976 to provide an average of 40% cover among the 1979 survey sites. Phyllis Faber began the monitoring work with a survey of vegetation, soil salinity, and habitat quality. One condition of the Bay Conservation and Development Commission permit was to establish a self-sustaining cordgrass habitat for waterfowl and shorebirds. In 1981, 5 years after the dikes were initially breached, tidal flows were inadequate for meeting this goal The upper part of the marsh was not developing the natural sinuous channels of the lower part (Gahagan and Bryant et al. 1994). Subsequently, new channels were created on the perimeter of the marsh to increase tidal action. The following year cordgrass had increased to more than 380,000 plants from the 1979 count of 1600 (Williams et al. 1988). The cordgrass is now more than 0.9 m tall, and cover continues to advance.

By year 10 of the project, additional species of plants began to appear. Between 1992 and 1993, 16 endangered California clapper rails and 10 species of fish were observed in both the constructed and natural marshes. Observations of both marshes indicate they are also used as fish nursery sites.

In 1992, Williams and Associates noted that a fully productive habitat at Muzzi Marsh is limited by the rate of sedimentation in the tidal marsh plains. Data suggest that it may take 60 years for the site to reach equilibrium (Goodwin and Williams 1992). Natural development of channels in the upper part of the marsh is slow. The high elevation of the marsh may preclude the development of sloughs (Gahagan and Bryant et al. 1994). The changing topography of the channels is a problem. Pools that attract fish for hiding and breeding are less abundant in the constructed marsh than in the reference marsh, thereby limiting the functions of the former.

Humboldt Bay Mitigation Marsh

Lesson: If functions of a destroyed habitat are to be replaced "in kind," hydrologic characteristics of the mitigation site must be equivalent.

The construction of Woodley Island Marina in Humboldt Bay, California (see Fig. 1.1a), required that mitigation be provided to replace lost shoreline, subtidal, and intertidal habitats. Construction of the marina also destroyed eelgrass beds and a population of piddock clams. The 3.5-ha site chosen for mitigation is located 5.2 km east of the marina at the upper end of Freshwater Slough Channel. Since 1875, the former salt marsh had been used primarily as pasture and secondarily as a sawmill pond. Restoration of the site consisted of dike breaching in December 1980.

To determine whether the restored marsh provided functions equivalent to those of the destroyed habitats, a 14-month study was done (Chamberlain and Barnhart 1993). Between July 1981 and October 1982, data were collected on the abundance, species diversity, spawning, rearing, and residence time of fish. Samples were collected from the adjacent channels, in the marina, in the restored salt marsh, and in an undisturbed (reference) marsh. Sampling methods included fixed channel nets, ichthyoplankton net, otter trawl, beach seines, and drop traps.

Not surprisingly, variations occurred between the constructed site, the control site, and the restored site. Variation in salinity was greater and mean salinity significantly lower in the water adjacent to the mitigation marsh. Likewise, salinities fluctuated more in the marshes than in the adjacent channels, particularly in the mitigation marsh, a shallow-water retention area (Chamberlain and Barnhart 1993).

English sole accounted for 50% of the total otter trawl catch in all areas combined, 65% in the channel adjacent to the marina (Eureka Channel), 44% in the channel adjacent to the control marsh (Eureka Slough), and 5% in areas near the mitigation marsh (Freshwater Slough).

Chamberlain and Barnhart (1993) concluded, "Even though a successful intertidal salt marsh was established, it did not adequately mitigate the lost habitat from construction of the Woodley Island Marina." The restoration created an upper estuarine habitat that primarily benefited juvenile and euryhaline fishes (Chamberlain and Barnhart 1993). It did not provide the subtidal and intertidal habitat lost at the marina site. Eelgrass flats and piddock clams were not replaced. The habitat at Woodley Island Marina was a relatively stable, subtidal marine embayment, beneficial to English sole and Dungeness crab (two commercially important species).

In addition, Chamberlain and Barnhart concluded that the choice of reference sites limited equitable comparisons because hydrologic and physicochemical characteristics were markedly dissimilar. They recommended that more acreage than that lost should be provided because of the time required to establish a fully functional system and because of the uncertainty of attaining equal habitat quality; they further recommended that data on both the construction and restoration sites should be provided before any alteration of either site was done.

Salmon River Estuary

Lesson: Subsidence of diked wetlands can slow restoration.

Located along the north-central Oregon Coast, the Salmon River estuary is part of the Cascade Head Scenic-Research Area established by the U.S. Congress in 1974. The long-range goal of the U.S. Forest Service was to return the area to a system free from the influences of humans. Before restoration, the 21 ha former high salt marsh had been diked for pasturage. Restoration began with partial dike breaching in 1978. No channel excavation or slope contouring was done, and native plant species were not transplanted into the area. Restoration depended solely on the natural establishment of salt-marsh vegetation.

Monitoring took place over 10 years, beginning in 1978. Mitchell (1981) compiled the first data before dike removal and analyzed wetland attributes over the next 2 years. Characteristics of vegetation, site elevations, salinity, soil texture and organic content, sedimentation, and erosion were studied. In 1988, Frenkel and Morlan (1990) surveyed the site, adding to and summarizing the body of data.

Intrusion of salt water initially eliminated the established pasture cover. Two years after breaching, salt marsh vegetation began to appear on the mostly bare site. As upland pasture communities were replaced by salt marsh communities, the diversity of plant species generally diminished. Pickleweed and saltgrass became established on the seaward area; sedges (*Carex lyngbyei*) dominated the inland edge. These three species dominated the restored marsh by 1984 and by 1988 accounted for more than 70% of the vegetative cover (Frenkel and Morlan 1991). Over the 10-year restoration period, elevation of the creek banks generally increased, and the creeks became narrower and deeper. Primary productivity in the restored area was nearly double the productivity of the undiked control and pasture areas and is thought to be typical of a developing ecosystem (Frenkel and Morlan 1991).

Although data were not recorded for numbers and species, juvenile fish, birds, and mammals were observed using the restored estuary 11 years after dike removal. This would indicate that a naturally functioning, low salt marsh ecosystem is developing. To assess its progress accurately is difficult. Even the existing undiked marshes used as reference wetlands have been indirectly disturbed by human activities.

At the time of breaching, the restoration area had subsided about 35 cm because of buoyancy loss, compaction, and organic soil oxidation (Frenkel and Morlan 1991). In the first decade, accretion of sediment had increased the elevations 5–7 cm in the lowest areas and 3–4 cm in the higher areas. To achieve the goals of the U.S. Forest Service may take five decades or more (Frenkel and Morlan 1990). Changes in wetland characteristics continue. Long-term monitoring will provide data necessary to determine whether some intervention is appropriate and desirable.

Frenkel and Morlan (1990) have observed that the objective of the U.S. Forest Service to return the Salmon River Estuary to its natural condition, free from the influences of humans, was *only partially* attained, because of the difficulty in assessing pristine conditions. Re-creation of entire communities of organisms, plants and animals included, that are closely modeled on those that occur naturally (Jordan et al. 1988) seems to be a realistic general goal for restoration. In these terms, restoration of this site on the north shore of the Salmon River can be considered successful (Frenkel and Morlan 1990).

Gog-Le-Hi-Te Wetland System

Lesson: Development of wetland functions is not always a simple progression.

The Gog-Le-Hi-Te Wetland System was constructed by the Port of Tacoma on the upper part of the Puyallup River as mitigation for filling an existing wetland habitat. Construction began in 1986 with a breach in the river dike, connecting the 3.9-ha site to an existing estuary, and subsequent transplantings of sedge.

The U.S. Army Corps of Engineers permit required monitoring over a 5-year period (1986–1990). Systematic sampling of sedimentation, vegetation, fish, birds, infauna, and epibenthic zooplankton was done in 1987 and 1990 (Shreffler et al. 1990, Thom et al. 1991). By the second year of sampling, the populations of infauna and epibenthic zooplankton had appeared to stabilize, and no further collection was done (Thom et al. 1991).

A primary objective of the project as established by resource agencies was to allocate 50% of the new wetlands for use by juvenile salmon. The remaining habitat areas were designed to support waterfowl, shorebirds, raptors, and small mammals (Shreffler et al. 1990). Although use by salmon appears to have increased since 1987, the goal of 50% use may be limited by the increasing sedimentation in the channels. The 1990 increase in the numbers of salmonid caught reflects one particularly large catch and may not represent a long-term trend (Thom et al. 1991). The 1990 monitoring showed that nine species of fish occupied the tidal channels. This represented a slight decrease in the number of species present in the previous 2 years but was comparable to numbers found during the 1986–1987 surveys.

The number of bird species in the estuary was 118 in 1990. This is a 50% increase in the number present in 1986. As many as 500 individual birds were counted using the site on a particular day. Some data suggest that the system has reached its carrying capacity; the number of species (approximately 36) present during the summer months has remained stable over the monitoring period (Thom et al. 1991).

The density of sedges increased at a dramatic rate between 1988 and 1990. The initially transplanted 48,000 shoots had increased to more than 420,000 plants by 1990. During this same period, area cover decreased from a high of 16.5% in 1987 to 7.4% in 1990. Changes in shoot density and the amount of aboveground and belowground biomass were used to estimate the rate of development of the transplanted sedge (Thom et al. 1991). A plant survey in the same year listed 57 plant species found in 1 or several of the 10 habitat categories. Of these, 56 species have colonized naturally.

One problem in evaluating the success of Gog-Le-Hi-Te is the lack of comparable surveys and the absence of good nearby reference sites. Simenstad and Thom (1996) observed that the lack of specificity in the performance criteria will haunt interpretation of the project's success forever. It is evident from the continuing changes in the hydrology and species composition of the marsh that considerable time is required to establish equilibrium in wetland restorations. Scouring and deposition of sediment are increasing in the tidal channels, and use by juvenile salmon may diminish. As a target resource, diminished use by salmon may be seen as a failure to fulfill the terms of the mitigation agreement. Indeed, because this is the only emergent marsh habitat now available to migrating juvenile salmon in the Puyallup River estuary (Shreffler et al. 1990), some intervention may be necessary to increase the subtidal habitat. Other problems noted are the accumulation and removal of rubbish, the seasonal extremes that can threaten emerging ecosystems, and the dangers of extrapolating linear models of development to wetland systems.

The period of monitoring specified in the original permit has elapsed. However, the Fisheries Research Institute continued sampling by using Gog-Le-Hi-Te as a U.S. Army Corps of Engineers Waterways Experiment Station demonstration site. Simenstad and Thom (1996) have presented an overview of the site development, with 7 years of data on physiography, vegetation, and animal use and special attention to the issue of performance criteria.

Pacific Coast Terminals Salt Marsh Compensation

Lessons: 1. Coarse soil, low supplies of nitrogen, and grazing may slow growth of transplanted plants.
2. Trial plots may be useful when restoration techniques are limited.
3. Historic dumping may impair the potential for restoration.

Construction of a new storage tank facility at the Pacific Coast Terminals in British Columbia was mitigated by the development of a salt marsh habitat. The project was required by the national fish habitat policy of the Canadian Department of Fisheries and Oceans (Williams 1993). Salt marsh restoration is relatively new in this area. The results of a 1-year monitoring program suggested that existing strategies for estuarine restoration may be ineffective for habitat of this type (Williams 1993).

Intertidal sediment dredged from the construction site was used as fill for the new marsh. The site was graded in May 1993. After placement of the dredged materials, the area was 0.5 m higher than in 1991. Although not previously used as a dominant species for transplanting, saltwort (*Glaux maritima*) was harvested from approved sites and transplanted as cores during May 1993 (Williams 1993).

Monitoring was done at the mudflat, the compensation marsh, and an adjacent salt marsh (as control). Data were collected on soil density, pH, and electrical conductivity. Analysis of fertility included measurements of total carbon and nitrogen and of amounts of available nitrate and ammonia. Periodic observations of growth were recorded, and photographs were taken.

Results showed significant differences between the control and the constructed marshes. In the constructed marsh, soils were much coarser, soil pH was more acidic, and electrical conductivity was lower than in the control marsh or mudflat (Williams 1993). Saltwort and pickleweed were the predominant colonizing species. However, at the end of the first growing season, plant growth was so poor that coverage was not determined. One factor limiting initial plant growth was the high carbon-to-nitrogen ratio in the constructed marsh. Successful early establishment of transplants was severely affected by intense grazing by Canadian geese. According to G. Williams (personal communication), the site was historically used as a dumping area for both sawdust and sulfur. Although the layer of sawdust was removed, chunks of sulfur remain on the site. Generation of acids from the oxidation of sulfides may be the cause of poor plant growth. Nevertheless, both saltwort and pickleweed were considered the most appropriate species for transplantation (Williams 1993).

Subsequent recommendations for monitoring emphasized the need to establish trial plots in which a variety of remedial measures are used. Some suggestions for improvement included creation of tidal channels, soil amendments for greater nutrient availability, and revised harvesting and planting schedules. The need to develop an ecological perspective to habitat compensation was proposed as a future objective (Williams 1993).

CHAPTER 3

Problems in Restoring Wetlands

3.1 PROBLEMS DURING IMPLEMENTATION

The basic problems in restoring or creating coastal wetlands concern hydrology, substrate, and biota. Of these, hydrologic problems are documented most often, partly because they are easier to see and partly because the effects on the entire system are dramatic.

3.1.1 Hydrology and Topography

Providing both proper tidal flushing and topography is critical to restoration of coastal wetlands. Marsh plants are extremely sensitive to the degree of stagnation (anoxia) and salinity of soil. Even a 10-cm mistake in elevation can prevent the desired marsh plants from growing well, if the hydroperiod (frequency and duration of tidal inundation) differs from what is needed. In addition to appropriate microtopography, suitable circulation patterns are required. In areas that receive too little tidal current, prolonged anoxia may develop, thus reducing plant growth; whereas in areas with too strong a current, erosion may occur, thus removing transplanted marsh species. Either improper planning or mistakes in implementing restoration plans can result in unsuitable hydrology. During construction, the over- or under-excavation of intertidal wetland sites can create major problems, because species are so sensitive to microtopographic differences.

Unexpected events that cause sedimentation and unexpected changes in tidal access during construction of a site can result in erosion and deposition, neither of which aids the establishment of transplanted vegetation. A common problem occurs in diked wetlands that have been removed from tidal flushing for many years; the substrate subsides, and elevations are lowered. Simply breaching the dikes does not restore the wetlands to their predike condition, however. The salt marsh of Oregon's Salmon River Estuary is an example. Because of subsidence, the restored site is mostly low-marsh habitat that may take a century or more to accrete enough sediments to bring the topography back to its prediked condition (Frenkel and Morlan, 1990; cf. Salmon River Estuary case study in chapter 2).

Too much accretion causes problems where tidal channels are cut into semitidal or nontidal areas (cf. Warm Springs Marsh case study in chapter 2). Such excavated areas often sediment in, especially for sites dredged deeper than the downstream channel that feeds them. Sedimentation rates have increased following development activities in the watershed. Onuf (1987) reported a 40% decrease in the low-tide volume of Mugu Lagoon after floods transported sediments to the lagoon mouth. As sediments fill intertidal channels, tidal flows become sluggish, and sand builds up across the ocean inlet.

Corrective measures are available for areas of subsidence (e.g., dredge spoils may provide suitable fill) and areas of accretion (e.g., maintenance dredging after the fact), but these measures prolong the time the ecosystem is disrupted and increase the likelihood of invasion by exotic species (cf. sections 2.3.3 and 3.4). Hence, it is best to plan with the knowledge that topography and hydrology may change during the development of an ecosystem. It is essential to predict the processes that will control water salinity, water flow, deposition of sediments, and erosion. When sedimentation events can be predicted, it may be possible to compensate by overexcavating channels that are likely to accrete sediments (P. Williams, Philip Williams Associates, personal communication).

Perhaps the most obvious hydrologic or topographic error is that constructed marshes rarely "look like" natural marshes. The constructed marshes always lack the complex networks of small tidal creeks, and broad, flat marsh plains are compressed or absent (cf. Muzzi Marsh case study in chapter 2). Bulldozers often create a collage of habitat types that rarely would abut one another, for example, steep-sided islands and deep, straight channels (cf. Hayward Regional Shoreline Marsh case study in chapter 2). Isolated habitats can pose serious problems for biota that need transitions between habitats (cf. sections 2.4 and 3.3). We recommend using natural marshes as models of the topographic complexity and habitat connections needed by native biota.

3.1.2 Substrates

The quality of the substrate and water often differs for natural and constructed wetlands. The substrate of excavated sites may become acid when exposed to air, as accumulated sulfides are oxidized to sulfuric acid (cf. Pacific Coast Terminals Salt Marsh Compensation case study in chapter 2). Excavation can inadvertently uncover toxic materials if the former wetland had been used as a landfill. Restoration sites may also be downstream of contaminated waters. In urban areas, marshes can accumulate trash, which smothers vegetation. Such locations will not support high-quality wetlands.

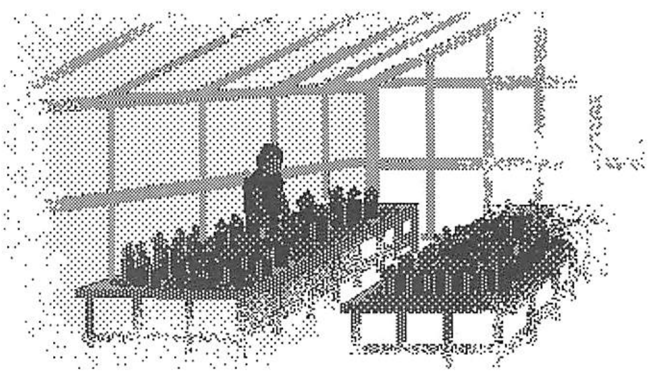
Soil texture. Natural marsh soils are fine-textured, whereas those of constructed marshes are often coarse, because the site of the constructed marsh often consists of dredge spoil deposits or upland fill. In chapter 5, we discuss the shortcomings of sandy soils in restored lower and upper intertidal marshes. The supply and recycling of nutrients are impaired when coarse-grained sediments are used to grow salt marsh vegetation (Langis et al. 1991). Water absorption, impoundment, and retention are impaired when sandy soils are used instead of clay-rich marsh sediments. Salinity regimes may also be affected if the substrate drains too readily (cf. section 5.2.4).

Amount of organic matter. Organic sediments are a basic feature of natural wetlands. It has been suggested that organic matter influences nearly every aspect of the functioning of wetland ecosystems: by changing sediment porosity and water-holding capacity, by influencing nutrient dynamics, by altering the growth rates and nutrient content of plants (cf. Pacific Coast Terminals Salt Marsh Compensation case study), and by influencing the species composition and abundance of invertebrates associated with the sediments. The microbes, plants, and animals in turn affect the rate of accumulation of organic matter in wetland sediments. For most wetlands, the development of organic sediments occurs over periods that are closer to centuries or millennia than to years or decades. In three regions of the United States—North Carolina (Craft et al. 1986, 1988), Texas (Lindau and Hossner 1981), and Southern California (Langis et al. 1991)—comparisons indicated that the levels of organic matter and nitrogen in sediments were lower in constructed and restored wetlands than in the natural reference wetlands (Table 3.1).

Soils of all of the constructed marshes had less than half the organic matter of natural marshes. Low levels of organic matter may impair microbial activities (nitrogen fixation, nutrient recycling, sulfate reduction).They may also influence faunal density and composition, not only through trophic linkages but also by altering the compactness of sediments and the ability of burrowing fauna to colonize the sediment (cf. section 5.1).

3.1.3 Biota

Several kinds of biological problems have been recognized in attempts to restore Pacific Coast marshes. These include limitations on transplantation, disconnected habitats, invasions of exotic species, and inadequate food chain support.



Transplantation. Attempts to grow and transplant native halophytes have led to several recommendations (Table 3.2). Because of the shortage of natural marshes, sources of transplants for restoration sites are scarce. Removal of plants from a region's wetland reserves is not advisable. Hence, plants should be salvaged whenever vegetation is being destroyed, so that they can be transplanted. It may be necessary to build interim nurseries to hold plants until restoration sites are available. In the San Diego Bay project, an intertidal nursery was built in 1983, planted with salvaged cordgrass, and allowed to grow for about 4 years. Mere construction of nurseries is not enough, however. The cordgrass nursery was accidentally destroyed in 1990, when the adjacent area was being bulldozed to expand habitat for cordgrass marsh. Another salvage effort failed when blocks of high-marsh vegetation were stockpiled and irrigated with fresh water for several years. Upland weeds invaded as the salinity of the soil declined, reducing the quality of the soil for restoration efforts. For successful salvaging of coastal vegetation and sod blocks, a long-term plan and a maintenance program for the nurseries are required.

Perhaps the most common problem with transplantation in Southern California is hypersaline soils. The average salt content of lower marsh soils of natural marshes is more than 40 ppt, and only a flood or a major discharge from a reservoir can lower the salinities to seawater or brackish levels. Marshes that have been diked become even more saline, and some of the highest salinities develop in dredge spoils. At San Diego Bay, the average salt concentration of newly excavated dredge spoils was 80 ppt on initial tidal influence; values dropped to 55 ppt in the first 2 months despite lack of rainfall and runoff. However, levels did not decline significantly over the next 4 months. Transplantation of cordgrass may need to be delayed until tidal flushing lowers salt concentrations to at least 50–60 ppt.

Table 3.1. Organic Carbon and Total Nitrogen in Natural and Constructed Marshes in Three Regions of the United States

		Organic Carbon (%)		Total Nitrogen (mg/kg dry weight)	
Location	**Marsh Age (yr)**	**Natural**	**Constructed**	**Natural**	**Constructed**
Texas	1	0.3–1.1	0.1	227–588	95
North Carolina	10–15	0.6–8.6	0.6–1.8	364–1,680	322–924
California	4	2.0–2.5	0.1–1.1	1,740–2,270	870–960

Note: Based on data from Lindau and Hossner 1981 (Texas), Craft et al. 1988 (North Carolina), and Langis et al. 1991 (California). Marsh Age = years between marsh construction and sampling.

Table 3.2. Considerations in Establishing Tidal Marsh Vegetation in Southern California

- Soil salinities should be lowered initially to facilitate halophyte establishment.
- An excavated site may need several months for tidal flows to leach accumulated salts and lower substrate salinity.
- Fall and winter are good times to transplant salt marsh vascular plants.
- Prolonged periods of low salinity will stimulate exotic plant invasions.
- Most halophytes will not colonize readily (e.g., cordgrass, Zedler 1986) and will need to be seeded or planted.
- Most salt marsh plants will establish from seeds and from cuttings that are rooted in a greenhouse.
- Seed germination is reduced and delayed by high salinities, as well as low moisture.
- Cordgrass (*Spartina foliosa)* seeds need to be ripened before planting; germination rates are highest when seeds are stored in the refrigerator in fresh water for 2–3 months and then germinated in fresh water.
- Each halophyte has a narrow range of tidal conditions that favor its growth; it is best to plant species at the elevation where it thrives in the nearest natural marsh.
- Cordgrass grows well in shallow tubs (kiddy pools) and can be propagated readily from rooted cuttings. However, permits are essential for collecting plants.
- Genetic diversity of transplants cloned from one or a few culms will be low; the consequences of low genetic diversity are not clear, but a diversity of starter plants (including plants grown from seed) is recommended for vegetative propagation efforts, while taking care not to contaminate local gene pools.
- Local stock should be used in all marsh restoration projects, using growers who can guarantee the source of their materials.
- A grower should be contracted to provide materials at least a year in advance of site availability.
- Pickleweed (*Salicornia virginica)* outcompetes cordgrass for nitrogen (Covin and Zedler 1988); hence, it should not be planted where cordgrass is planned.
- Pickleweed is a good invader if there is a seed source, but planting will speed canopy coverage.
- Glasswort (*Salicornia subterminalis)* grows slowly from seed but should be grown in tall pots to accommodate deep roots.
- Shoregrass (*Monanthochloe littoralis)* grows from cuttings and seeds and has established well at Marisma de Nación (San Diego Bay).
- Salt marsh bird's beak (*Cordylanthus maritimus* ssp. *maritimus)* has very specific planting requirements (section 2.4), and special permits are needed for this endangered species.
- Pilot plantings are recommended along transects perpendicular to shore; plant species at the elevation where initial survival and growth are best.
- Fertilize plantings experimentally to determine what limits growth and spread.
- Use irrigation for upper-marsh sites, especially where substrates are coarse.
- Monitor plantings for insect damage and small mammal herbivory; September to November is a likely time for small-mammal grazing, as the marsh soil is less often wet and there is less forage in the upland.

Note: Based on Zedler, 1984.

Disconnected habitats. When habitats are constructed as islands, the lack of terrestrial corridors can reduce or eliminate natural movements of some animals, even flying insects. Predatory beetles were not initially present at San Diego Bay's dredge spoil island (Chula Vista Wildlife Reserve), and their absence seems to be responsible for the extensive attacks by scale insects on cordgrass transplants (cf. section 5.1). Where beetles abound in tall cordgrass of natural marshes and on the mainland surrounding the Bay, scale insect outbreaks may be kept in check by predation. Islands of high-marsh habitat at another San Diego Bay restoration site (see Fig. 2.5) provided suitable habitat for reintroduction of an endangered plant, the salt marsh bird's beak (cf. section 2.4). However, only 20% of the plants were pollinated (B. Fink, PERL, personal communication). Without a seed bank and with insufficient pollination to produce new seeds, annual species cannot persist. Bee pollinators need marsh-upland transition habitat for their ground nests and for alternative sources of nectar that feed the bees when bird's beak is not in flower.

Exotic species. Too often exotic species—both animal and plant—become established in constructed marshes (cf. Hayward Regional Shoreline Marsh case study in chapter 2, and section 3.4). Other than the knowledge that exotic species take up space and reduce the potential for the development of native species, we have little information on how these aliens change food webs and wetland functioning. A Japanese mussel *(Musculista senhousia)* invaded the constructed marsh at San Diego Bay in large numbers but was only rarely present in the natural reference wetland (Scatolini and Zedler 1996). Because many Pacific Coast estuaries already have large numbers of introduced invertebrates, researchers and others are concerned that moving transplant cores from one wetland to another (e.g., San Francisco Bay to Elkhorn Slough) will bring unwanted species along with the desirable plants (M. Silberstein, Elkhorn Slough National Estuarine Research Reserve, personal communication).

Food chain support functions. The few data available indicate that considerably lower abundances of invertebrates or vastly different species of invertebrates are present in constructed marshes than in reference wetlands. Scatolini and Zedler (1996) examined the epibenthos of San Diego Bay cordgrass marshes and found similar species but greatly reduced densities in the 4-year-old constructed site. Cammen (1976a, 1976b) reported significant differences in the infauna of constructed wetlands along coastal North Carolina. At a 1-year-old marsh, the dominant taxa were insect larvae, whereas polychaetes were dominant in the natural wetland. Total densities and calculated secondary production also were markedly lower than in the natural system. Similar observations were made at a 2-year-old site. Sacco et al. (1994) revisited the 2-year-old site when it was approximately 15 years old and found a 10-fold increase in densities and a high similarity with the infauna of natural marshes. Studies of an upland site that was graded to support *Spartina alterniflora* found that the constructed site had fewer nematodes, ostracods, harpacticoids, and oligochaetes than natural marshes (Moy 1989, Sacco 1989, Moy and Levin 1991).

Species composition does not always become more similar through time. Moy (1989) reported a divergence in composition over a 2-year period between a constructed marsh and another marsh receiving the same source water as the created site. Sacco (1989) evaluated the composition, density, and trophic structure of infauna in coastal North Carolina marshes 1–17 years after their construction. Although variable, the data showed that the six constructed marshes had similar faunal components and trophic groupings (deposit feeders, suspension feeders, and carnivores) but that densities were uniformly lower than in natural marshes.

A study of the food chain at Tijuana Estuary (cf. section 6.3) was conducted in 1993–94 to provide better advice for projects that seek to improve or create habitat for estuarine fishes. Fish support is a major requirement of several proposed mitigation projects in Southern California (e.g., San Dieguito Lagoon and Batiquitos Lagoon, section 7.2). The degree to which fish are supported by algae that grow in channels and by vascular plants or algae that grow in the salt marsh has significant implications for mitigation projects. If the fish depend on the marsh for food, then mitigation credit is suitable for large areas of intertidal marsh adjacent to subtidal channels.

3.2 SALINITY PROBLEMS

Southern California estuaries are small and isolated, and annual rainfall events (timing of rainfall, storm intensity, and streamflow) are all highly variable (cf. section 1.1). The norm is hard to define, but the region is considered semiarid; average annual rainfall is about 30 cm. Small watersheds are typical of most of the region's coastal streams; hence, inflows of fresh water occur in pulses. Under natural conditions, streams most likely had minimal flow in summer. During the cool season (November–March), which is the rainy season, estuarine waters become ephemerally brackish, although coastal springs may have created small areas of more persistent brackish conditions.

3.2.1 Reductions in Salinity

Virtually all municipal water in Southern California is imported and stored in reservoirs; Northern California and the Colorado River are the major suppliers. The release of water as street or irrigation runoff or its discharge as treated effluent alters the hydrologic regime of coastal rivers and downstream estuaries. There is often too much fresh water. And, because estuarine inlets in Southern California have a tendency to close, tidal flushing may be interrupted for months or years at a time (cf. section 2.1 and Los Peñasquitos Lagoon case study in chapter 2). There is often too little salt water to sustain the marine character of the coastal wetlands. During the nontidal intervals, fresh water can accumulate, and several biological changes can occur. Current research on the effects of reductions in salinity is inadequate for setting specific limits on the release of wastewater. However, the studies clearly show the need to consider the detrimental effects of deliberate discharge of fresh water to Southern California's coastal wetlands.

Because the coastal strip is almost fully developed, urbanization is moving inland, where it is more expensive to discharge wastes to ocean outfalls. Thus, it has been proposed that the wastewater from these newly developed inland areas be treated and discharged to coastal streams for reuse in irrigation downstream throughout the long dry season. The California Regional Water Quality Control Board (San Diego Regional Water Quality Control Board 1988) projected that 38–114 million liters of treated wastewater might be discharged each day to each of 10 coastal rivers over the next 25 years. Not all of the flow would reach coastal wetlands, but discharge of wastewater would exceed irrigation demands, at least in the wet season.

Untreated wastewater flows into the United States from the City of Tijuana, Baja California, which includes many neighborhoods that do not have sewer lines. Flows of wastewater were about 49 million liters/day of raw sewage to Tijuana River and Tijuana Estuary from about 1986 to 1991 (Seamans 1988, Zedler et al. 1992). Both long-term and catastrophic assaults on the coastal habitats have produced a highly stressed ecosystem. During 1993, catastrophic sewage spills in excess of 64 million liters/day caused beaches in the area to be closed for more than 220 days (C. Ricks, San Diego Association of Governments (SANDAG, personal communication). Tijuana wastewaters have high concentrations of pollutants, because of both low per capita water consumption and fewer controls on contaminants from industry. In fall 1991, the raw sewage was diverted to San Diego's sewage treatment plant during off hours. However, the collector system overflows during rain events. Even construction of the new border sewage treatment plant (due to operate in 1997) will not prevent winter flows of raw sewage.

An occasional source of excess fresh water is reservoir drawdowns. After major flooding occurs, water must sometimes be released from reservoirs to sustain flood protection. El Capitan Dam discharged water for several months after floods in January and February of 1980. Flows of fresh water to the San Diego River estuary continued through the summer (Zedler and Beare 1986). In 1993, Rodriguez Dam discharged fresh water well after the 12 days of continual rain that occurred in early January of that year.

Other inflows of fresh water occur where agricultural irrigation and street drains funnel flows from irrigated fields or lawns into nearby coastal wetlands. Finally, spills from storm drains or sewer systems occasionally fill wetland channels, especially during heavy rainfall and runoff. Tijuana Estuary has had continual inflows of raw sewage from the upstream metropolis of Tijuana (Seamans 1988, Zedler et al. 1992). Since 1993, these sewage flows have been collected and diverted to San Diego's treatment plant, so that flows to the estuary are now restricted to times of floods that exceed the capacity of the interceptor.

3.2.2 Increased Freshwater Inflow

Under natural conditions, many wetlands in Southern California would receive little fresh water except during streamflow pulses in winter after the occasional rain. There would be an extensive period of little or no flow during the long dry season. Changing the hydrology from intermittent freshwater pulses (e.g., four to five major flow events in 15 years) to continuous flows reduces the salinity of the estuarine waters and sediments, and the biota respond when major shifts occur from saline to brackish or fresh conditions.

Prolonged periods of low salinity convert salt marsh habitat to brackish marsh dominated by cattails (*Typha* spp.) and bulrushes (*Scirpus* spp.; Zedler and Beare 1986, Beare and Zedler 1987). Prolonged freshwater influence can also eliminate marine fish and invertebrates (cf. section 2.1). Fish and invertebrates, which are well adapted to seawater, are killed during natural floods, but populations can reestablish themselves after marine waters again dominate the estuary. Where streamflows persist because of discharges, the natural channel assemblages are eliminated.

At Tijuana Estuary, fish and macroinvertebrate species have been less diverse since wastewater inflows became substantial (beginning about 1986). The numbers of individuals collected in our monitoring programs have dropped by an order of magnitude, and large clams are rare. Young-of-the-year are present, so we know that larvae are available to settle, but they rarely survive to reproduce (Nordby and Zedler 1991).

In order to determine whether a reduction in salinity alone (and not contaminants in the raw sewage) can eliminate species, various experiments have been done with fish and invertebrates. The conclusion is that low salinity causes mortality, especially of molluscs (Nordby, Baczkowski, and Zedler, unpublished data). More detailed experiments on California halibut showed that growth of juvenile fish is reduced when salinity is low and that the effects are greatest on the smallest individuals (Baczkowski 1992). Dry-weather discharge of fresh water is thus considered a significant factor, above and beyond the problems caused by materials carried by inflowing water.

The effects of wastewater discharge can be reduced by releasing water in pulses rather than continuously (Zedler et al. 1994), and experiments on both fish and molluscs showed reduced damage when exposure to low-salinity water was via pulses (Nordby, Baczkowski, and Zedler, unpublished data). We have done mesocosm experiments to show that wastewater wetlands could be constructed to retain water and discharge it in pulses. We hypothesized that such wetlands would have an improved ability to remove nutrients from the water column, much as East Coast freshwater tidal wetlands support more nutrient uptake and higher plant productivity than wetlands with more constant water levels. Twice-a-day impoundment and discharge not only improved the capability of wetland mesocosms to remove nutrients but also enhanced uptake of heavy metals (Busnardo et al. 1992, Sinicrope et al. 1992). Thus, we think that the problem of reduction in salinity can be decreased by using pulsed-discharge wetlands upstream of the estuary (see section 3.2.4). However, this is only a means of reducing adverse effects, not a way of eliminating them.

3.2.3 Eutrophication and Contaminants

Urban runoff and sewage spills introduce a variety of unwanted materials to coastal bodies of water. Pathogens are of particular concern in urban areas, and frequent problems arise in coastal areas, such as San Diego County. During 1992, the ocean outfall for the sewage treatment plant at Point Loma (City of San Diego) spilled more than 378 million liters/day of treated effluent into shallow ocean waters. The spill continued for several weeks. The threat of contamination by pathogens led to the closure of all nearby shores, including those in San Diego Bay. The region's favorite recreational beaches were closed, and tourism was no doubt affected. San Elijo Lagoon appears to have a leaking sewer pipe under it, which allows pathogens to enter the water column (Dodge 1994, Matkovits 1994). When the mouth of the lagoon is dredged open for tidal restoration, nearby beaches are contaminated and must be closed to the public (R. Gersberg, Graduate School of Public Health, San Diego State University, personal communication). Sewage flows from the City of Tijuana present the largest health threat to Southern California beaches. Fecal coliforms have been monitored at Tijuana Estuary during recent years, and during the rainy season levels are approximately the same as those of raw sewage (Gersberg et al. 1994). Adjacent beaches are usually closed during rainfall events.

Nutrients, detergents, oils, pesticides, fertilizers, heavy metals, and other toxic materials are also present in urban runoff. Once contaminants reach the downstream estuary, they may become concentrated near the inflows or diluted and carried to the ocean. At times and in places, the flows are impounded by sand bars that block tidal flow. Contaminants adhere to sediments and accumulate. These become hot spots for contaminants (Trindade 1988, Gersberg et al. 1989) and high biological demand (Nordby 1988, 1989).

Heavy metals are abundant in wastewater entering Tijuana Estuary. Samples of surface water have contained mean levels of 69 ppb of cadmium, 55 ppb of chromium, 281 ppb of nickel, and 321 ppb of lead (Gersberg et al. 1989). Estuarine sediments act as a sink for heavy metals, containing up to 1.7 ppm of cadmium, 25 ppm of chromium, 14 ppm of nickel, and 59 ppb of lead. Concentrations of heavy metals are particularly high where sewage waters are impounded (Trindade 1988). Heavy metals and other toxic materials enter wetland food chains and concentrate in organisms, but the extent of the contamination is not clear. No studies of biological concentration in Southern California's coastal wetlands have been published, although some sampling of the tissues of various organisms has been done (S. Goodbred, U.S. Fish and Wildlife Service, personal communication).

Eutrophication (additions of nutrients and associated responses) as a result of wastewater discharge is only generally known. Mexican sewage contains more than 25 mg/L of nitrogen and more than 10 mg/L of phosphorus. We know that productivity of vascular plants in Tijuana Estuary is nitrogen limited (Covin and Zedler 1988) and that macroalgal blooms are stimulated by wastewater inflows (Fong et al. 1987, 1993). Salt marsh vegetation is more productive in years of major floods, a consequence of better growth under the combined influence of lowered soil salinity and increased nitrogen availability (Zedler, Nordby, and Kus 1992).

3.2.4 Recommendations

Year-round recycling of municipal water would eliminate the problem of excess freshwater flows to coastal wetlands. Greater reuse of imported water would also reduce diversion of water from the San Francisco Bay delta, where freshwater inflows are needed to sustain the biota. Similar concerns exist for the Owens Valley and the Colorado River (Zengel et al. 1995).

Total recycling. Treated wastewater can be used to irrigate crops, but Southern California does not have sufficient agriculture near the sites of wastewater production. Hence, for reuse to solve the problem, most likely we would need to use treated wastewater for drinking and other domestic purposes. When fresh water is brought in from Colorado, it includes treated wastewater that has been discharged in other states. It is made potable by further purification in water treatment plants. However, drinking of wastewater effluent that has been produced and treated within California is permitted only if the effluent is first passed through a groundwater aquifer to reduce the potential for virus transmission. More research is needed on the safety and acceptability of total recycling, and better methods are needed to remove pathogens and ensure the safety of reused water.

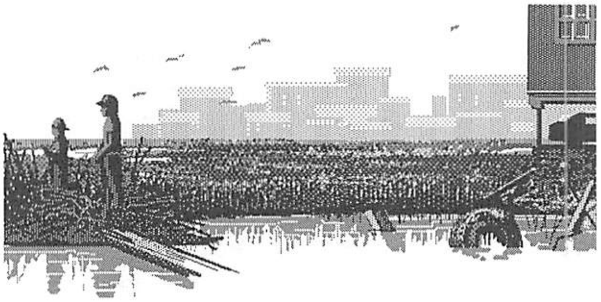
Using excess fresh water to create wetland habitat. Treated wastewater could be used to construct and maintain freshwater wetlands that would subsidize habitat for estuarine birds while marsh vegetation and soils improve water quality entering the estuaries. Bulrush wetlands have high capacity for nitrogen removal, with more than 90% reduction of total nitrogen at hydraulic residence times of 5–6 days (Gersberg et al. 1984, Busnardo et al. 1992). Constructed wetlands are also capable of removing both bacterial and viral indicators of pollution with a removal efficiency of nearly 99.9% for poliovirus (vaccine strain; Gersberg et al. 1987). Sinicrope et al. (1992) reported the removal of substantial amounts of the heavy metals present in wastewater. A recent study of the fate of trace metals in a constructed wetland in Hayward, California, found similar results for the removal of copper and zinc (Gregg and Horne 1993). However, concentrations of nickel and lead increased to greater than those present in influent wastewater and are thought to be the result of aerial deposition. Reducing the concentrations of these metals at the treatment plant or at the source may be necessary (Gregg and Horne 1993).

Because augmented inflows are detrimental to Southern California's estuaries, constructed wetlands would need to be engineered to discharge treated wastewater in pulses. A brief influx of fresh water would minimize adverse effects (e.g., reduction in salinity) on the estuary. Recent experiments showed that rates of removal of heavy metals and nitrogen were highest during a hydroperiod of twice-daily impoundment and discharge (Busnardo et al. 1992, Sinicrope et al. 1992). With this regimen, denitrification rates may be enhanced, leading to removal of additional nitrogen from the system. Also, problems with mosquitoes might be fewer because of the more rapidly flowing water.

The potential for using constructed wetlands to manage wastewater in Southern California needs to be explored. Pulsing the discharge of treated wastewater could reduce the impact of the reduction in salinity and improve the quality of effluent entering the coastal wetlands. However, total recycling has greater potential for eliminating the problem of too much fresh water.

Pacific coastal wetlands require a regional approach to research and management (section 4.2). In Southern California, the amount and timing of discharges must be managed to maintain native vegetation and associated fauna. It is not sufficient for managers to worry only about the loss of fish and shellfish habitat, because endangered species are often jeopardized by loss of wetlands. Management models cannot be derived from data on regions where plants and animals are more tolerant of brackish water.

Controlling contaminants. There are no programs for consistent monitoring of contaminants in coastal bodies of water. Monitoring of pathogens is intermittent, and efforts do not include all wetlands. Beaches used for swimming and surfing are the usual targets, even though people also swim and wade in lagoons. A great deal of research needs to be done to assess these problems and to develop solutions to prevent point-source discharges that contain contaminants and to provide remediation at sites that are already contaminated.



3.3 PROBLEMS OF URBAN WETLANDS

Urban wetlands are important to city dwellers, who gain an occasional glimpse of native habitat. They are also essential to native species that depend on specific wetland habitats. In Southern California, endangered species compete for space with a variety of urban uses, such as highway widening, dredging for recreational boating, and filling for housing and commercial uses. There is not enough coastal land to fulfill the nation's policy of "no net loss" of both wetland habitat and functioning and still allow unrestricted urban growth.

Even when urban wetlands are set aside as reserves, habitat degradation continues. The hydrologic effects are discussed in section 3.1. Locally important effects include trampling and associated disturbances caused by an ever-growing human population that enjoys open space.

3.3.1 Visitors

The proximity of an admiring public raises the issue of how many visitors wetlands can tolerate, especially when visitors bring their pets. Allowing visitors to experience wetland habitats helps the public appreciate and support wetlands. Proponents of free access by visitors argue that people can safely approach birds in urban wetlands. However, no data are available that indicate whether the birds that continue to use urban wetlands are the same ones that would be there without urbanization or whether the birds that remain are just the species and individuals that can tolerate human disturbance (cf. Hayward Regional Shoreline Marsh case study in chapter 2). We can never know how sensitive the natural populations are, because they are no longer natural.

The immediate effects of great number of visitors include disturbances caused by noise and movements, trampling of vegetation, and predation by cats and dogs. The presence of humans in and around coastal wetlands is disruptive to birds (Powell 1993). Experiments at PERL showed that shorebirds have a range of sensitivities to a person walking near their feeding site (Table 3.3). The great blue heron is the most skittish, whereas coots are relatively tame. In order to plan for the highest biodiversity, the needs of the most sensitive species must be accommodated. A 60-m buffer would be reasonable to keep visitors from disturbing the great blue heron. Currently, the California Coastal Commission requires a 30-m buffer, and that can be narrowed if the applicant shows that a broader buffer is not required. A 7.5-m buffer was approved for developments around Los Cerritos Wetlands in the Los Angeles area.

Powell (1993) showed that Belding's Savannah sparrows (*Ammodramus sandwichensis beldingi*) are disturbed by the presence of visitors at about 30 m during the birds' mobile, nonterritorial phase. However, when they are defending their nesting territories, the birds can be approached closer. Whether this impairs the bird's nesting or feeding activities is unknown. Others (Jungius and Hirsch 1979) have shown that birds' heart rates increase before flight, indicating that animals can be stressed before their behavior so indicates.

3.3.2 Trampling

The effects of trampling are significant on a local scale, especially at the upper margin of the salt marsh. Several high-marsh vascular plant species have narrow elevation ranges where tidal inundation is infrequent (*Atriplex watsonii*, glasswort, salt marsh bird's beak). Because this upper-marsh band is seldom muddy, it is a choice location for informal walking trails. Repeated trampling eliminates the more sensitive species first (e.g., *A. watsonii*, glasswort) leaving the tougher plants (shoregrass, saltgrass) to dominate until all vegetation is denuded. One entire patch of the endangered salt marsh bird's beak was obliterated at Tijuana Estuary by unauthorized foot traffic; the nightly migration of undocumented workers from Mexico to the United States via the Tijuana estuary was responsible. The entire estuary has fewer than 15 patches of bird's beak, so this loss was substantial. When surveillance of migrants was intensified, trampling was greatly reduced, and the bird's beak patch reestablished itself.

The effects of trampling can be far-reaching. At Tijuana Estuary, both vehicle and foot traffic have nearly eliminated the vegetation that formerly stabilized the barrier dune. Without vegetation, sand is blown and washed inland, filling estuary channels and reducing the tidal prism along with sediments from the watershed. At Agua Hedionda Lagoon, the excavation of tidal basins as part of a mitigation project (Kel-Cal 1985) has inadvertently

Table 3.3. Effects of Human Disturbance on Wetland Birds

Species	Behavior Change Min.	Behavior Change Max.	Flight Min.	Flight Max.	No. of Days Observed
American coot	12	14	8	11	4
American widgeon	40	43	24	37	4
Cinnamon teal	26	26	20	21	3
Eared grebe	11	15	6	8	5
Snowy egret	20	29	15	20	5
Common egret	27	30	14	18	3
Green heron	17	17	5	11	6
Great blue heron	0	0	55	62	4
American avocet	24	27	17	23	4
Black-necked stilt	15	24	14	21	4
Short-billed dowitcher	6	6	5	5	2
Killdeer	8	17	8	15	6
Willett	8	17	8	11	5

Note: Data taken by Pam Beare (PERL) indicate distances she could approach before birds changed their behavior and before they flew. For some species, flight occurred before a behavioral change was noted. Numbers of observations were small and varied by species. Distances estimated by eye; minimum and maximum distances rounded to the nearest m.

increased abuse of adjacent natural salt marshes by riders on off-road vehicles. In this case, the mitigation site was not excavated low enough to be fully tidal; hence, it is a dry salt pan that invites vehicle traffic, which moves from the salt pan to the adjacent pickleweed marshes when the marshes are not too muddy. Substantial damage to the natural vegetation has occurred. This mitigation project not only did not achieve its habitat goal (replacement of a pickleweed marsh), it also catalyzed degradation of the nearby natural marsh.Vehicle traffic denudes the vegetation, reduces habitat quality, and compacts the soil. Revegetation is difficult in hard soil, and disking to loosen the soils can enable colonization by exotic species of plants (cf. section 3.4).

Denudation of dune vegetation at Tijuana Estuary did not produce a crisis until a natural sea storm washed the sand into the tidal channels. Closure of the estuary occurred a year later, in 1984, when rainfall and streamflow were too low to push sediments through the mouth of the estuary. An 8-month nontidal drought then decimated populations of cordgrass, invertebrates, and, eventually, the light-footed clapper rail (Zedler, Nordby, and Kus 1992). The sea storm compounded the effects of increased sedimentation and trampling. Small-scale dredging was necessary to remove sediments and restore tidal flushing in the short term; a major dredging and restoration program is needed to ensure continual tidal flows in the long term.

3.3.3 Lack of Buffers

In Southern California, there are no native trees to act as natural buffers between coastal wetlands and adjacent upland developments. There is no easy way to screen wetlands from lights, noise, and motion, because fencing is costly and areas adjacent to wetlands are too saline for most tall woody plants. Even if these measures were cost-effective, it would be inadvisable to develop up to the extreme high tide line. Such developments would not only be inundated during storm swells, but also be doomed to inundation in future decades as sea levels continue to rise.

There are additional reasons to provide buffer zones between wetlands and urban development. Broad expanses of open space are available in natural wetlands, and the transition zone between wetland and upland habitats supports the wetland biota.

Although it may not be obvious except on extreme high tides, the transition zone between wetland and upland provides high-tide refuges for mobile species that do not like to remain in deep water. At San Quintin Bay, 330 km south of San Diego, the broad transition-zone habitat is used by wetland birds as a resting site during high tides. Wetland insects move to high ground when the tides inundate the marsh plants. From the terrestrial side, small mammals, lizards, and snakes burrow or hide in the upland and the transition zone and move into the wetland at low tide. The importance of such wetland-upland linkages has not been quantified, but we suspect it is great on the basis of two restoration problems in San Diego Bay (section 2.3).

The endangered salt marsh bird's beak requires more than the wetland habitat where it grows. This high-marsh species appears to rely on buffer zones in at least two ways. As an annual, it must set seed in most years to sustain the population, and pollination is essential (Parsons 1994). The plant's pollinators are terrestrial insects, that is, solitary bees that nest in soil above high tides. It also requires small open spaces for seedlings to become established and grow through the marsh canopy. Large open areas are unsuitable, because the plant is a hemiparasite that grows best in association with various host plants. The disturbances that provide small patches of open space may depend on the actions of small mammals (Cox and Zedler 1986). Unfortunately, little information is available on the kind and amount of habitat necessary to maintain the appropriate populations of burrowing mammals.

3.3.4 Recommendations

Southern California's urban wetlands are extensively and intensively disturbed. Visitors disturb wetland biota and trample habitat. Trampling can eliminate species with narrow elevation ranges. The effects of trampling of dune vegetation can be magnified by natural sea storms and rising sea level. Recommendations include the following:

- Visitors should be kept on paths that are clearly marked and located in places that prevent adverse effects on sensitive species.
- Visitors' activities (noise, lights, pets) should be regulated to minimize adverse effects on wildlife.
- Buffers are needed to distance wetlands from developments. They are also needed to function as high-tide refuges and as habitat for upland species that play important roles in the adjacent wetland (pollinators, mammals that open small patches in the high-marsh canopy).
- Further research is needed to help set the widths of buffers and determine allowable activities within buffer zones. Until such information is available, it makes sense to use the broadest buffer zones that can be justified. Data on great blue herons suggest that 60 m is appropriate.

3.4 PROBLEMS WITH INVASIVE EXOTIC PLANTS*

Invasive exotic plants threaten coastal salt marshes, both by invading adjacent mudflats and by displacing native vegetation.Several situations promote their invasion, including enhanced dispersal, substrate disturbance, and

*Modified from Zedler 1992.

hydrologic modifications (especially reduced soil salinity). A common feature is that short-term alterations in environmental conditions initiate long-term problems that are not easily solved; management strategies should therefore focus on prevention.

Records of the arrival and early spread of exotic plants are rare, and some species introduced by humans are accepted as natives. Although it may not seem fair to distinguish species carried by humans from those that arrived via the feet, feathers, or guts of other well-known dispersal agents (notably birds; Vivian-Smith and Stiles 1994), the distinction does emphasize the fact that humans have had an inordinate influence on the world. If we knew how propagules arrived and what conditions allowed their spread, we might be able to predict spread and suggest control measures. Indirect evidence of when exotic species first became abundant can be obtained from pollen profiles in wetland sediments (Mudie and Byrne 1980). Adobe bricks include seeds of weeds that were present at the time of the bricks' production. Herbarium collections and species maps may indicate times and species that were historically rare and places where the species later became abundant, and new floras indicate expansions of exotic species in recent years (Rejmánek and Randall 1994).

3.4.1 Adverse Effects

Exotic plants that can grow lower in the intertidal zone than native plants can reduce the area of mudflat available to shorebirds and some invertebrates of economic interest. A species with greater tolerance to inundation may 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. Invasion of intertidal flats in Great Britain by nonnative cordgrasses (section 3.4.3) has reduced feeding habitat for native wading birds.

When exotic plants invade vegetated areas, they may displace native species. In Southern California, the invasion of the New Zealand mangrove, *Avicennia marina*, into the habitat of native cordgrass (*Spartina foliosa*) is considered detrimental to both plants and birds (cf. section 3.4.3). A few mangroves were planted in Mission Bay, San Diego; their escape prompted concern that the trees would displace cordgrass, which is the preferred nesting habitat and nesting material of the light-footed clapper rail, and that the trees would also provide roosting places for raptors, which readily prey on rail chicks (P. Jorgensen, California Department of Parks and Recreation, personal communication). An active eradication program followed, with annual visits to pull out mangrove seedlings by hand.

The indirect effects of changes in vegetation are not well known. Although 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 insects are limited to individual families or genera of plants; others are restricted to a single species or to a single life stage (e.g., flowers or seeds) of a host plant. These same dependencies are present among the insect inhabitants of coastal wetlands, yet wetland insect communities have received little attention (C. Nagano, U.S. Fish and Wildlife Service, personal communication), 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 species of salt marsh grass (*Distichlis spicata*) for larval feeding.

Changes in vegetation are not always considered entirely detrimental. Recent and rapid changes in vegetation are taking place along the estuaries of the Swan and Canning rivers near Perth, Western Australia (Zedler et al. 1987; cf. 3.4.4). *Typha orientalis,* a cattail, is rapidly expanding into native salt marsh, where it replaces *Juncus kraussii* and other native species (Pen 1983). Like 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 acceptable (R. Atkins, Waterways Commission, Perth, personal communication). Likewise in England's Poole Harbour, exotic cordgrass is valued for its ability to control shoreline erosion (Gray 1985). The managerial opinion is that a little is good, whereas a lot is not (A. Gray, Institute of Terrestrial Ecology, personal communication).

Managers are generally more concerned about losses of animal habitat than about changes in composition per se. Threats to rare and endangered plant species and major changes in plant type are the exception. Salt marsh bird's beak is an endangered plant species that occurs at the landward edge of intertidal salt marshes in Southern California (section 2.3), where its habitat is often disturbed, and weedy species sometimes invade. Coulter goldfields *(Lasthenia glabrata)* is a rare plant in Southern California (Ferren 1985). At the remnant population in Los Peñasquitos Lagoon, it occurs in close association with an exotic annual *(Cotula coronopifolia;* J. Boland, PERL, personal communication*)*. Invasion of Florida's brackish wetlands by cajeput trees (*Melaleuca quinquinervia;* section 3.4.3) 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 salt marsh bird's beak. Tree removal was required after the invasive nature of the exotic plant was publicized, and concerns for the native vegetation were voiced. Damage control is invariably an expensive response to poor planning.

In at least one case, an exotic has become the overwhelming dominant plant of a major coastal wetland.

The discovery (Spicher and Josselyn 1985) that the dominant cordgrass in Humboldt Bay is an exotic from Chile (*Spartina densiflora*) rather than the California native (*Spartina foliosa*) has prompted concern that this invader might displace the native cordgrass over a broader geographic area (M. Josselyn, San Francisco State University, and P. Kelley, California Department of Fish and Game, personal communication).

3.4.2 Conditions Promoting Invasion

The conditions that allow invasion by exotic wetland plants can be determined by considering what prevents the establishment of these plants in the least disturbed wetlands. Dispersal, disturbance, temporary relief from environmental stress, and prolonged changes in the environment all contribute to invasions.

If habitats are suitable for species that are not present, then the only limiting factor is the availability of propagules. Deliberate or accidental introduction of seeds or plants (e.g., planting of mangroves in Mission Bay) will then enable establishment. 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 (e.g., disking that creates suitable microsites for seed germination) may enable establishment of exotic species. If the abiotic environment is 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 allow invasion (e.g., invasion and persistence of *Typha domingensis* in San Diego River Marsh; cf. section 3.4.4). If the abiotic environment is unsuitable, then a persistent change in environmental conditions may enable invasion (e.g., invasions of *Rumex crispus* after prolonged impoundment of fresh water and long-term deposition of sediments from upstream erosion at Los Peñasquitos Lagoon; cf. section 3.4.4).

If two or more limiting factors are removed, invasion moves from the possible to the probable. Such was most likely the case for the early invaders whose seeds traveled to new lands in cargo ships, fell on shores that were being developed into ports, moved up newly constructed navigation channels, or encountered estuaries where hydrology was being altered by changing patterns of land use upstream in the watershed. Multiple factors appear to be responsible for the rapid advance of *T. 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 for the establishment of *Typha* seedlings, but the combination enables seedlings to become established in the drainage channels; thereafter, *Typha* invades the native marsh by vegetative expansion of adult plants (Zedler, Paling, and McComb 1990)

A number of invasions in coastal wetlands have been described recently, and although the specific events that caused the establishment and spread of the exotic plants are undocumented, the probable causes can be inferred from the association with certain changes. Some invasions after deliberate or accidental introductions of species, some after major hydrologic changes, and others are common wherever substrates are disrupted. Each invasion may have multiple causes.

3.4.3 Deliberate or Accidental Introduction

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

The primary management concern is the loss of mudflat habitat for shorebird feeding (Goss-Custard and Moser 1988). 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." Herbicides have been used for control in northwestern England (Long and Mason 1983). Some diebacks have occurred, for which the causes are unclear. Only about half of the maximum population persists in Poole Harbour (Gray 1985).

Spartina species are presently of concern in California. *Spartina densiflora* is a caespitose (bunch) grass that dominates the intertidal salt marsh of Humboldt Bay and has begun to invade San Francisco Bay. Until recently, it was thought to be an ecotypic variant of the native *Spartina foliosa*. However, Spicher and Josselyn (1985) showed that it is a Chilean species, probably carried in ballast with the timber trade. In San Francisco Bay, *Spartina densiflora* occurs slightly higher in the intertidal zone than *Spartina foliosa* and outcompetes the native pickleweed (*Salicornia virginica*) (Josselyn and Buchholz 1984). Its spread in San Francisco Bay has been summarized by Josselyn and Buchholz as follows: The species was introduced to Creekside Park in 1976 and by 1984 had expanded to a range 14 km in diameter. 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 seawater, and are produced prolifically some 2 months earlier than the seeds of *Spartina foliosa*. These characteristics and its high productivity make it a threat to the *Salicornia virginica*–dominated marsh and the upper part of the habitat of *Spartina foliosa*. However, it is not expected to replace the native pickleweed in the upper part of the marsh, where salinities can be high. Information on its effect on animal populations is lacking, but these authors state that "until the evidence supports that *Spartina densiflora* is not detrimental, all efforts should be made to control its spread to other locations in the bay."

Spartina alterniflora occurs in San Francisco Bay at the mouth of the Alameda Creek Flood Control Channel and about 3 km to the south. The reason for its introduction and its date of arrival are unknown (Spicher and Josselyn 1985). Aberle (1990) reported that this species was introduced to Willapa Bay, Washington, in the late 1800s 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 hunters of waterfowl. The species is now considered a major pest species in Puget Sound, Willapa Bay, and San Francisco Bay, and managers are seeking control methods (Aberle 1990).

Other exotic plants are potential threats to California wetlands. Examples are the following:

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 and then release them after a stimulus such as fire (Drew and Schomer 1984). Trees were planted along the periphery of Tijuana Estuary by homeowners in Imperial Beach, San Diego County. Its potential for salt marsh invasion was not considered because local managers were unaware of its pest status elsewhere. Landscapers installed drip irrigation and planted the trees in imported soil. The trees established well but were later removed after the management concerns were recognized.

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

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 valued horticultural attributes (ease of transplantation with use of short branches that readily root at the nodes and rapid vegetative spread) are the same characteristics that confer weediness. Eradication is difficult, because small pieces of plants regenerate and seeds germinate in the disturbed sites where adult plants have been pulled. At Los Peñasquitos Lagoon, San Diego County, the Department of Parks and Recreation has attempted a control program, which will be successful only if maintenance is continual to remove new sprouts and seedlings (W. Tippets, Department of Parks and Recreation, personal communication). This species has invaded a wide range of coastal habitats, including strand and dunes (Williams and Williams 1984), chaparral (Zedler and Scheid, 1988), bluffs, and salt marsh (e.g., Los Peñasquitos Lagoon).

3.4.4 Invasions After Hydrologic Modifications

A salt marsh exists as a community in part because its component plant species can tolerate high salinity, rather than because they require hypersaline (salt >34 ppt) soils. Lowering soil salinities allows most species to grow better, but if low salinities persist, brackish marsh vegetation can invade.

In many of California's coastal wetlands, tidal flushing is discontinuous. The ocean inlets of smaller wetlands and those with relatively small watersheds tend to close when sand accumulates during the summer (cf. Los Peñasquitos Lagoon case study in chapter 2). When filling and accretion of sediment reduce a lagoon's tidal prism, the frequency and duration of closure are likely to increase. If closure is followed by periods with streamflows that are sufficient to reduce the salinity of the lagoon but insufficient to break through the sand barrier, the wetland will experience a prolonged period of freshwater flooding, which will lower soil salinity (section 2.1.2). Under such conditions, the following exotics are likely to invade:

Cotula coronopifolia (brass buttons) is a herbaceous, succulent perennial from South Africa. It is common in both fresh and saline wetlands. Its seeds germinate at salt concentrations of 10 ppt 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 downstream of Mexican sewage spills.

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) indicated that salt concentrations less than 10 ppt allow recruitment from seed; the plant is a perennial but does not reproduce vegetatively. At Los Peñasquitos Lagoon, San Diego County, *R. crispus* has become a conspicuous component of the higher marsh, sharing dominance with *D. spicata*, the native saltgrass. Its abundance has increased significantly in recent years, after prolonged periods of closure and inundation by local runoff.

Typha domingensis (cattail) is a widespread dominant plant of brackish wetlands. Its invasion of the San Diego River salt marsh occurred after the 1980 flood and subsequent, prolonged reservoir discharge. Experimental studies showed that few seeds germinate at salt concentrations greater than 20 ppt and that seedlings required several months of low salinity soils to develop rhizomes and persist in tidally inundated soils (Beare 1984; Beare and Zedler 1987). Field monitoring of soil salinities indicated that the San Diego River Marsh was brackish (in this case, salt concentration <10 ppt) for most of 1980 (Zedler and Beare 1986). There was a strong positive correlation between streamflow and soil salinity, which supported the cause-effect relationship between reservoir discharge and changes in salinity in marsh soil. The abundance of cattails also correlated with variations in streamflow. The expansion rate was greatest in 1980, when seedlings became established over most of the intertidal salt marsh. Biomass dropped in 1981 and 1982, which were low-flow years. During 1983, a year of high streamflow, the cattail population flourished through vegetative regrowth (not establishment of seedlings), because soils were again brackish for part of the growing season (Zedler and Beare 1986). Since then, the population has declined to relatively low levels. Under experimental conditions, rhizomes resprouted even after being held at a salt concentration of 4.5% (about 1.3 times that of seawater) for an entire year (Beare 1984). Reinvasion would be likely under prolonged freshwater influence, because rhizomes can expand rapidly, even if soil salinities are not low enough to allow germination of seeds. The records for *T. orientalis* in Australia (see following) support the generality of the low-salinity invasion phenomenon.

T. orientalis is native to eastern Australia but not to Perth, Western Australia, where the climate is similar to that of Southern California and where *T. orientalis* has invaded and replaced native salt marsh plants in the estuaries of the Swan and Canning rivers. There, street drains reduce the salinities of marsh soils (Pen 1983), and experiments have shown that the combination of lowered salinity and substrate disturbance is responsible for invasions (Zedler, Nordby, and Griswold 1990). Tolerance to salinity increases with the age of the plant, and vegetative growth can occur in conditions far too saline for the establishment of seedlings. The seedlings are not good competitors, so disturbed soils must coincide with low salinities for seedlings to survive (Zedler, Nordby, and Griswold 1990). Thus, invasion is restricted to street drains where soils are disrupted and salinities are nearly fresh for unnaturally long periods. Once seedlings develop rhizomes, the plant becomes more salt tolerant and more competitive. A single seedling can establish a clone and expand vegetatively into the native marsh. It is hypothesized that similar mechanisms govern invasion of clonal exotic plants in wetlands in Southern California.

3.4.5 Invasions After Disruption of Substrate

The transition from wetland to upland is often marked by an increase in the species richness and an abundance of exotic plant species. The origins of exotic species, according to Munz (1974), include Europe (*Polypogon monspeliensis*), Eurasia (*Bassia hyssopifolia, Salsola iberica*), Chile (*Cortaderia atacamensis*), South Africa (*Mesembryanthemum* spp.), and Australia (*Atriplex semibaccata*). (Abrams [1951] states that *M. nodiflorum* probably arrived in California before Europeans did). The exotic plants are not necessarily aggressive plants in their home country. In England, for instance, *P. monspeliensis* is valued as a rare plant, and its habitats are actively managed to maintain populations (A.J. Gray, personal communication).

These exotic species mark areas where the coastal soils have been disturbed by agriculture or horticulture, dumping, sediment plumes, street drains, disposal of spoils from dredging operations, trampling, or vehicle use. The species 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 grow where tidal debris is deposited and marsh vegetation is smothered, so long as the elevation is high in the intertidal zone (e.g., extreme high water). These peripheral invaders obviously are somewhat tolerant to salt, but their tolerance to inundation appears limited. Their restriction to bluffs and the periphery of salt marshes suggests either a low requirement for salt or poor competitive ability in nonsaline soils. In Southern California, *Atriplex semibaccata* is a common dominant plant of the marsh-upland transition zone (Ferren 1985, Cox and Zedler 1986, Zedler and Nordby 1986).

The list of alien weeds would probably be lengthened by further studies of the high marsh and the transition zone to upland throughout California. This transitional area is the most poorly known habitat of coastal wetlands. Most of its area has already been destroyed, and what remains has been badly disturbed.

3.4.6 Recommendations

California's coastal wetlands are a limited natural resource that is jeopardized by invasive exotic species. In general, the higher marsh and buffer zones are more susceptible to invasion than the lower marsh, because few species can tolerate both frequent tidal submergence and hypersaline soils. Combinations of disturbances, such as hydrologic changes plus soil disruptions, improve the opportunities for germination of seeds and establishment of seedlings. Exotic species can spread rapidly at the expense of native plants, but the conditions that promote invasion cannot always be discovered. Once established, naturalized exotic plants are difficult, if not impossible, to eradicate. As coastal landscapes are modified, propagules of exotic species are accumulating, increasing the potential for invasions of exotic species.

A native plant conservation ethic is needed. Native plant species should be allowed to dominate California's natural coastal wetlands, and recently introduced species should be controlled, if not eliminated. The rationale behind this judgment is straightforward: California's coastal wetlands are small and few, about 130 in the entire state. The remaining wetlands have been highly modified and severely reduced in area; common estimates are losses of 75–95%. 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 not be supplied by alien species.

Control measures become essential whenever the invader is considered noxious or the affected resource is highly valued. In California, all the coastal wetland habitats are highly valued, not only as native ecosystems but also as habitat for rare and endangered plants and animals.

The best time to control an invasive plant species is before it gains dominance. If caught early, the disturbance caused by attempts to control exotics will affect a smaller area, thus reducing chances for reestablishment of weeds. Experiences with weed control in wetlands are limited, and success stories are few. Efforts to eradicate *Chrysanthemum* from filled areas at Tijuana Estuary by disking only retained the disturbed-soil conditions that facilitated its dominance. Controlled burning to eliminate seeds and accumulated biomass is often ineffective (W. Tippets, personal communication) and, if done in the driest weather, may be hazardous to adjacent developments. Early diagnosis and treatment are essential, and use of hand tools rather than machines is desirable. When use of herbicides is necessary, spot application by hand is preferable to broadcast application. Local or regional regulations may require the application of herbicides within a wetland or stream corridor by licensed contractors only (Vanbianchi et al. 1994). Generally, use of herbicides is discouraged because of potential contamination of wetlands resources.

Invasive exotic plants should be controlled before they eliminate native wetland vegetation. Restoring native high-marsh vegetation will be difficult because seed mixes to enhance germination of native halophytes are unavailable from nurseries. Hence, gathering and sowing by hand are required; problems should not be left until large-scale operations are necessary.

Preventing excess discharge of fresh water can reduce problems. Controlling street runoff to salt marshes will help eliminate a major cause of weed invasion, namely lowered soil salinity. Upstream sediment traps are certainly needed; hookups to storm drains offer better protection for wetlands. When large drains enter estuarine channels, and connections to storm drains are not possible, controls at the source of the problem may have to yield to treatment of the effects. In such situations, it is important to maintain good tidal flushing.

Native plants should be used to landscape the buffer zone. Using native plants to landscape the transition zone is a simple way to reduce chances that horticultural species will invade the wetland. An educational program is needed to show that native species are esthetically pleasing, that planting methods are being developed, and that nurseries can produce adequate stock as demands arise. Desirable attributes of native plants for landscaping transitional habitats need to be publicized. For example, *Juncus acutus* (spiny rush) is an effective deterrent to cats and dogs; a border of these sharp-leaved plants is nearly impenetrable. *Malosma laurina* is drought and salt tolerant; it requires no care after initial establishment. Native insects prefer native plants over exotic plants such as *Carpobrotus edulis*.

The integrity of native sod should be maintained. Disking, tilling, and grading are not suitable methods of weed control; on the contrary, they propagate the problem. Instead, native vegetation should be encouraged to dominate the site. This is done by using small-scale soil preparation (patch rototilling or augering) and planting. At Tijuana Estuary, exotic species were quick to dominate sites prepared by disking, whereas native species were slow to colonize the sites. Disking also altered soil topography and impounded rainfall so that unnatural "furrows" of wetter soil developed in the high marsh, stimulating the establishment of exotic plants (B. Fink, PERL, personal communication).

Continual surveillance is required to control existing problems and avoid future expansions. It is overly optimistic to aim for eradication of species that are invasive and widespread; however, reducing the number of existing problem areas and preventing spread to new sites are reasonable goals.

Wherever the approach to a problem is uncertain, small-scale experimental approaches should be used first. As an example of a small-scale approach, we are experimenting with salt as a weed-control measure, seeking the dosage and timing of application that will both allow germination of seeds, establishment of seedlings, and growth and reproduction of native species, while preventing reproduction and spread of exotic species (Kuhn 1995; Callaway and Zedler, in preparation).

CHAPTER 4

Recommendations for Improved Planning

4.1 GOALS AND PERFORMANCE STANDARDS IN RESTORATION AND MITIGATION PROJECTS

Restoration and creation of habitat are ways to improve the status of natural biological resources. Sometimes these activities are undertaken solely for that purpose, and sometimes they are intended to mitigate damages to habitat, that is, to ensure that no net loss of wetland area or function occurs.. The distinction between these two types of projects is important: the former may not require specific performance standards, whereas the latter must be able to show that damages have been compensated. That is, quantifiable performance standards based on measurable attributes must be established, so that resource agencies can determine whether the mitigator has complied with the mitigation requirements.

The return of a habitat to some condition that existed previously is generally referred to as restoration. Because ecosystems are dynamic, no single period constitutes the previous state. Furthermore, no detailed records of pristine conditions (e.g., before European colonization of North America) are available, so it would be difficult to establish clear objectives. Restoration "targets" are often general, involving reintroduction of tidal influence, removal of fill or accreted sediments, reintroduction of native species, control of exotic species, and so forth. Attempts at restoration are sometimes paid for by resource agencies or nonprofit groups. Mitigation agreements provide another source of funding for restoration projects. Of concern is whether the restoration plan is designed on the basis of the best management practices or is governed by the needs of a mitigator. In the latter case, set acreages of specific habitat types may be imposed on a disturbed or degraded wetland in ways that are less than optimal from a resource perspective (Zedler 1996b).

The National Environmental Policy Act defines mitigation as avoiding, minimizing, rectifying, reducing, eliminating, or compensating for impacts on natural resources. The concern dealt with in this book is compensatory mitigation, that is, attempts to make up for damages to existing wetlands. Section 404 of the Clean Water Act regulates wetlands by requiring a permit for filling or disposal of dredge spoil and mitigation for unavoidable damages to wetlands.

Normally, projects that are not water dependent can be moved to uplands to avoid adverse effects on wetlands. However, some projects that are not water dependent are still permitted in wetlands in Southern California. For example, the City of San Diego is relocating and expanding an existing sewer pump station within Los Peñasquitos Lagoon. Objections to the pump station helped confine construction to an area of existing fill, avoiding salt marsh. However, the pipelines leading to and from the station are being cut through salt marsh. Likewise, highways and railroads are not water dependent, but Interstate Freeway 5, the Pacific Coast Highway, and the Santa Fe Railroad cross most of Southern California's coastal wetlands. The continual widening of these roadways further affects the remaining wetlands. In one such project along San Diego Bay, wetland was damaged by the widening of Freeway 5, a new freeway interchange, and a new flood control channel. Three endangered species were jeopardized.

4.1.1 Mitigation Issues Related to Goal Setting

The first items that need to be considered in developing criteria for mitigation are the location of the mitigation site, how much area is required, whether a former wetland can be restored or a new one must be created, and the quality of the resulting wetland. More detailed issues include the functions and values of the wetland that will be lost, the exact type of habitat to be provided, the time when lost functions are to be restored, and the performance standards that will be used to determine when the project is in compliance. Goals need to be appropriate for the region in question.

Restoration or creation? Federal mitigation policy (Environmental Protection Agency and U.S. Army Corps of Engineers 1990) recommends that restoration be given priority over the creation of new wetlands from upland. This policy makes sense in some places, such as prairie potholes that are drained and farmed and no longer function as wetlands. Restoration of former potholes is more desirable than excavation of potholes from natural upland. However, where the degraded wetlands still perform critical functions, this strategy is doubly damaging. First, the degraded site is altered without our knowing what critical functions were lost, and second, a net loss in wetland area occurs when alteration of the degraded site (still a wetland) allows damage to another wetland (Zedler 1996a).

In Southern California, loss of habitat is often mitigated by restoring areas that are modified but still functional wetlands; it rarely involves construction of new wetland habitats from nonwetlands. Thus, a net loss of wetland acreage occurs, and often habitat quality declines (Zedler 1996b). As stated in *A Citizen's Guide to Wetland Restoration* (Vanbianchi et al. 1994), "Nothing is gained if we destroy or damage one wetland to enhance another."

Mitigation ratios. Mitigators are often required to restore two to four times the area they damage. Part of the rationale for a mitigation ratio greater than 1:1 is that the compensatory habitat may have lower functional value than the damaged site. Another reason is the delay between the damage to habitat and the compensation for loss. The time for habitat replacement can be long (cf. case studies on Muzzi Marsh and Gog-Le-Hi-Te Wetland System in chapter 2).

The California Coastal Commission normally requires a 4:1 ratio for habitat replacement, but this ratio may not be sufficient when habitats of endangered species are concerned. Even where 4:1 ratios are used, they do not always sustain wetland area. When existing disturbed wetlands are simply modified, a net loss in wetland area occurs: if 1 ha is destroyed and 4 ha of existing, degraded wetland is remodeled, the net loss is 1 ha. Such practices do not fulfill the policy of no net loss of wetland area. Only the creation of wetland from nonwetland areas can replace lost wetland *acreage*.

Habitat quality. Restored or created wetlands that have lower functional value than the damaged site may not compensate for the loss, especially if what is damaged is habitat of endangered species (cf. Humboldt Bay Mitigation Marsh case study in chapter 2). Even if 3:1 or 4:1 compensation is required, a larger area of unusable habitat will not replace the functional value of 1 hectare that is critical to the endangered population. At the least, agencies should require assessment of the functioning of the mitigation site before and after improvements and "up-front" mitigation, with "success" achieved and documented before destruction of the development site.

Project timing. Projects are rarely planned with indefinite end points. Usually a specific period is allowed for compliance. Most often the compliance period is shorter than the time required for the restored or constructed wetland to achieve functional equivalency with the conditions that existed at the site before any damage occurred. Although 5 years has been a common time frame for many projects, longer monitoring periods are now being required.

Construction times are often lengthy, especially where extensive dredging is required. Excavation and disposal of dredge spoils are both time-consuming. Off-site disposal of fine sediments is also costly. At Batiquitos Lagoon, fine sediments are being dredged to return tidal flushing; these sediments are being buried in situ, by overexcavating the middle embayment, removing the underlying sand and using it for beach replenishment, and placing the fine material from the rest of the lagoon dredging project in the resulting hole. Obviously, this extends the construction period, and biota will be disrupted for several years.

Despite relatively short compliance periods, projects should be designed to last indefinitely, and changing environmental conditions should be taken into account. Global changes anticipated within 50 years include changes in temperatures, increases in the rate of rise of sea level, alterations in rainfall patterns, and increases in storm activity. No policies require consideration of such changes. For coastal wetlands, accelerated rates of rise in sea level need to be considered in long-term planning. Some of the sediments that are proposed for removal in mitigation projects may ultimately be needed to offset rises in sea level.

Transferability of techniques used in other areas. Restoration techniques developed outside one region may not be applicable in another. In Southern California, management issues usually concern endangered species, and the species are endangered because so much of their habitat has been destroyed. From a national perspective, these systems may be the most difficult to restore because (1) the endangered species have specific requirements; (2) little wetland area is left in the region, so there are fewer propagules to ensure dispersal and reestablishment; and (3) restoration sites are fragmented, hydrologically modified, lacking in buffers from urban activities, and often contaminated.

In a broader context, it is difficult to transfer findings across the country, because of differences in physiography and climate. Although there is no simple way to quantify differences, Pacific coastal systems are subject to catastrophic events that pulse fresh water and nutrients into the wetlands. The variability of annual flow volumes in streams in San Diego County is one of the highest in the nation (Pryde 1976). In reviewing mitigation proposals and promises, agencies need to recognize the unique attributes of Pacific Coast ecosystems. Models developed in wet climates that have relatively little interannual variation are least likely to be applicable to regions with low rainfall and streamflow and high interannual variability.

4.1.2 Goals for Restoring Southern California Coastal Wetlands

The need for a regional restoration strategy for coastal Southern California is described elsewhere (Zedler 1996b). Because so much of the coastal wetland habitat has been lost, it is urgent that the goal of all further modifications be to sustain biodiversity. A short-term, small-scale perspective would suggest that every mitigation project should have the goal of on-site, in-kind replacement. But when many of the mitigation requirements are for deepwater fish habitat, and where the only place available to create such habitat is shallow-water wetlands, mitigation agreements sometimes allow one habitat type to be remodeled into another.

A regional strategy for restoration of wetlands would include a detailed inventory of all remaining habitats (status report), analyses of what habitat types have highest priority for restoration (regional needs), inventory of plans and mitigation needs in relation to constraints (determination of opportunities), and a mechanism for matching mitigators' requirements with regional resource management needs. The first two elements of this strategy are being undertaken in the San Francisco Bay area (J. Collins, Western Society of Naturalists presentation, December 28, 1994), where most of the state's coastal wetland area occurs. The process being undertaken there should aid efforts in Southern California. Indeed, the State Resources Agency has called for an inventory of Southern California wetlands in 1995 (C. Denisoff, personal communication). Existing maps of wetland resources will be developed into a geographic information system to aid in summarizing extant habitat types. In addition, Ferren and colleagues (1994) have developed a habitat classification scheme for Southern California wetlands that will facilitate categorization and inventory.

Coastal wetlands support the following habitat types, and increases in all of these are suitable restoration goals: salt marsh, salt pan, transition zone to upland, tidal creeks and channels, mudflats, sandflats, dunes, dune slack, and beach. Expansions of the populations of the native plant and animal species that depend on each habitat are also suitable restoration goals. It is clear that sustaining biodiversity requires protection and expansion of habitat. What is not so clear is what goals have priority when there are so many priority needs.

4.1.3 Performance Standards

The California Coastal Commission is developing statewide guidelines for setting performance standards for mitigation projects (Z. Hymanson, personal communication). PERL and Philip Williams Associates are assisting in the effort. Ideally, restoration and creation of habitat should strive for functional equivalency with undisturbed wetlands. The questions are then what functions should be measured, whether simple structural attributes are good indicators of functional capacity, and what attributes should be measured to test for functional equivalency.

Functions are processes. Familiar functions are primary productivity, decomposition and production of detritus, and production of finfish and shellfish. These attributes have been the focus of hundreds of papers on wetlands of the Atlantic and Gulf of Mexico coasts and several in the Pacific Northwest. Measuring these functions is important where fish production is the most highly valued function of wetlands. Along the coast of Southern California, where there are no anadromous fishes, where wetlands are too small to support much shellfishing, and where endangered species are of great concern, other functions are more valued. Support of biodiversity is of overriding importance in Southern California, and several functions contribute to the maintenance of native flora and fauna:

- Maintenance of natural hydrologic regimes in both short- and long-time scales. Persistent tidal flushing, interrupted occasionally by small floods and more rarely by extreme floods, allows recruitment of marine fauna (which are intolerant of fresh water, Zedler et al. 1994) and flora, most species of which require lowered salinities for major recruitment (Zedler and Beare 1986, Callaway et al. 1990).
- Maintenance of habitats with suitable area, size, shape, and connectivity. Large areas of consistent tidal influence at Tijuana Estuary, with a low ratio of perimeter to area and a broad connection to the river floodplain, support at least 24 species considered sensitive in the region (Zedler, Nordby, and Kus 1992).
- Provision of buffers that protect wetlands from human activities in uplands, sustain habitat for upland species that are essential to wetland species, and allow room for the wetland to migrate inland as sea level rises (cf. section 3.3).
- Resistance to disturbances, that is, the ability of ecosystems to sustain natural processes and populations as habitats degrade in quality and decline in area. For example, tidal flushing sustains high soil salinities, which prevent invasion of most exotic species into the lower intertidal salt marsh (cf. section 3.4).
- Resilience, that is, the ability of the system to recover from short-term disturbances, both natural and anthropogenic. For example, epibenthic invertebrates (*Cerithidea californica* and various crabs) reinvaded Tijuana Estuary when tidal flows were restored after the estuary had been closed for 8 months.
- Maintenance of nutrient supply and retention capability. Natural marshes have sufficient nitrogen to sustain tall canopies of cordgrass but not excessive supplies that would generate massive algal blooms which would smother invertebrates.
- Persistence of vegetative cover that provides suitable nesting habitat and protection from predators.
- Small-scale patchy disturbances to allow recruitment of annual plants. Salt marsh canopies do not become too dense or too sparse to support species

such as the endangered salt marsh bird's beak, because the disturbance agents (small mammals, wrack deposition) are sustained at the appropriate scale.

Ecologists have not developed methods for assessing all these functions. Even methods of measuring primary productivity that have been tested repeatedly in eastern wetlands are of questionable use in Southern California (Onuf et al. 1978). Not only do succulent species have high biomass loss rates, making harvest methods inaccurate, but also the restricted status of the wetlands in Southern California suggests that harvesting is too damaging to be allowed as a regular assessment technique. No standard set of measures exists that must be made to assess the functional equivalency of restored and natural wetlands.

Because functions require more effort to measure than structural attributes, it is likely that performance standards will emphasize structure, that is, what there is at one specific time. Like a snapshot of the system, the presence and abundances of various species, the height of the canopy, and the cover of algal mats can all indicate the presence of various processes. Yet they must be recognized as indicators and not presented as proof.

Functional equivalency indexes. Earlier, a simple index of functional equivalency for the Connector and Paradise Creek marshes along San Diego Bay was published (PERL 1990, Zedler and Langis 1991). Only the best areas of the constructed marsh, i.e., the areas deemed most likely to function the same as natural marshes, were sampled. The index included 11 attributes, and the constructed Connector Marsh was less than 60% functionally equivalent to the natural marsh.

Such indexes are subject to misinterpretation and misuse. The comparison of the Connector and Paradise Creek marshes was considered a best-case rather than a worst-case or an average comparison, because the bare and average canopy areas of Connector Marsh were not sampled. Thus, it should not be concluded that the constructed marsh was 60% as functional as the natural marsh. Rather, the data suggested it was less than 60% functionally equivalent in the attributes sampled. Addition of other attributes might have increased or decreased the value. Also, the manner in which individual results are entered into an index affects its overall comparison. The extensive data base of Rutherford (1989) on invertebrate composition was lumped into one value, whereas nitrogen fixation rates were split into surface and rhizosphere values, thus emphasizing nitrogen supply more than invertebrates. The comparison was restricted to attributes measured in the same place at approximately the same times, although adding other measures had little effect on the index.

Before adopting such an approach as a performance standard, it would be important to determine whether data from the same wetland, sampled over time or space, would be within 80% to 90% similarity. Next, the sensitivity of the comparison to the manner in which attributes are included would need to be determined. Finally, it would be useful to explore the use of weighted averages, that is, giving more emphasis to some attributes than to others. There may be reason not to equate the contribution of surface nitrogen fixation rates with the support of the entire epibenthic invertebrate assemblage, for example.

Performance curves. Kentula et al. (1992) suggested using performance curves to describe the development of restoration sites. They proposed that over a period of a few years, the natural marsh would exhibit a relatively stable condition, whereas the constructed marsh would begin with low functional status and rise to some relatively stable level below that of the natural marsh. Simenstad and Thom (1996) have presented real data for several attributes of Gog-Li-Hi-Te in Puget Sound. Their results indicate much greater complexity: Some attributes of the constructed wetland increased, some decreased, and others changed unpredictably. Reference wetlands differ realistically, with equivalency partly dependent on which site is used for comparison.

The implication that both natural and constructed marshes have stable performance (horizontal lines) over short periods is debatable. Over centuries, a representative mean may be useful, but a short-term mean may bear little relationship to that mean. Thus, it is recommended that performance curves be developed simultaneously for natural and constructed wetlands. Although it would be cheaper for a mitigator to select 5-year data sets as the restoration target, the reference wetland may perform better or worse during the assessment period for the mitigation site.

Long-term monitoring data from Tijuana Estuary and San Diego Bay suggest an alternative way to track the performance of a constructed or restored wetland. Callaway and Zedler plotted the absolute condition of several structural attributes of natural marshes through time and the relative condition of the same attribute in the constructed marsh. For each year, the constructed system was compared with the reference site for the same year (Fig. 4.1). The results showed that no steady increase in structural equivalency occurred within the 8-year monitoring program of this 11-year-old wetland.

Performance standards for nesting by the light-footed clapper rail. Assessment of the performance of Connector Marsh as potential nesting habitat for the light-footed clapper rail suggests an additional approach that is appropriate for specific mitigation or restoration targets. Because the basic problem with this marsh is its coarse, inorganic substrate, which does not supply enough nitrogen for the growth of tall cordgrass, a functional equivalency index that emphasizes cordgrass density or mean height will not distinguish adequate and inadequate vegetation. The appropriate measure of habitat suitability here is an assessment of canopy architecture, specifically the height distributions (Zedler 1993). Criteria were

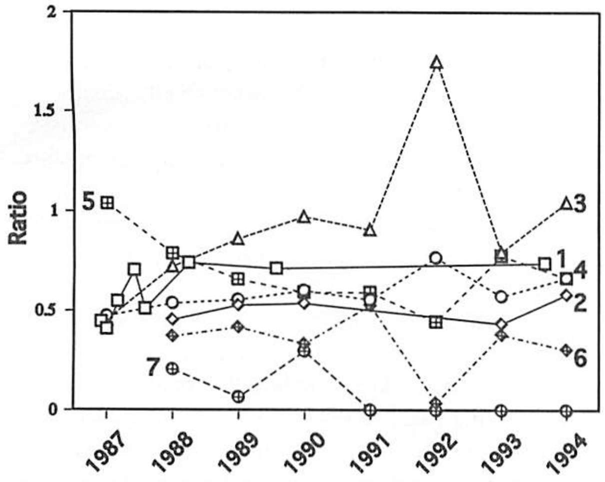


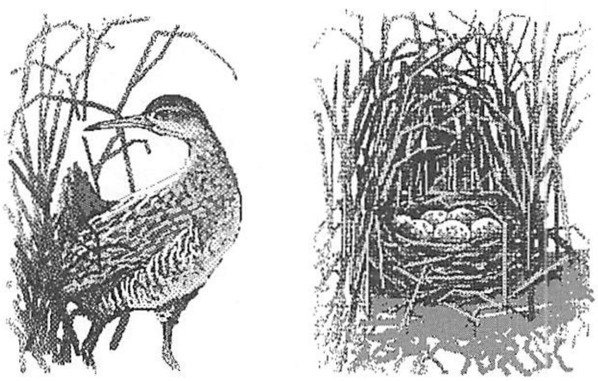
Figure 4.1 Changes in 7 attributes of sediment (solid lines) and cordgrass (*Spartina foliosa*, dashed lines) in a constructed wetland that was transplanted to cordgrass in 1985. Data from each year are expressed relative to the condition of the adjacent natural marsh. Values <1 indicate that the constructed wetland has less than the natural marsh. It is difficult to predict if or when the two marshes might become equivalent. 1 = sediment organic matter; 2 = sediment total Kjeldahl nitrogen; 3 = stem density; 4 = total stem length; 5 = mean height; 6 = number of stems > 60 cm tall; 7 = number of stems > 90 cm tall. Data of Callaway and Zedler (PERL, unpublished).

developed after a detailed analysis of canopy characteristics in various marshes with and without clapper rails sampled in various years. It considered the nesting habits of the clapper rail (elevation where nests are built, thickness of nests), tidal levels during the nesting season, and the main causes of nest mortality (inundation by tidal water and predation).

The proposed standards are that there be at least 100 stems of cordgrass per square meter, with at least 90 stems taller than 60 cm and at least 30 stems taller than 90 cm. These are to be present within a sampling area 20×20 m, randomly sampled with at least 10 circular quadrats, each 0.25 m^2 (Zedler 1993). The mitigation agreement (cf. Table 2.8) for Connector Marsh and Marisma de Nación calls for at least seven home ranges, each 0.8–1.6 ha, and each having at least 15% low-marsh and at least 15% high-marsh habitat (U.S. Fish and Wildlife Service 1988). Each home range must have at least one 100-m^2 patch of tall cordgrass.

Understanding how the habitat serves the species and comparing sites where the species does and does not nest are what led to the cordgrass performance criteria. Additional performance standards for clapper rail nesting sites should include adequate nitrogen supplies, tidal flushing, forage species, and adequate high-tide buffers. Once we work out the nitrogen fertilization regimen needed to sustain the tall vegetation, it should be possible to assess nutrient levels in the soil and inflowing water and predict the site's ability to grow tall cordgrass in perpetuity. Nesting of clapper rails at the Connector Marsh would provide assurance that tall vegetation, and not urban disturbances, has limited their use of the site to date.

Draft performance standards for salt marsh bird's beak. An approach similar to that used for assessing potential nesting habitat for clapper rails can be taken to set suitable criteria for reestablishment of salt marsh bird's beak. Several design criteria follow from ongoing work (see section 2.4.4). Soils should have a texture similar to that of natural marsh sites that continually support the species, and tidal flushing and elevations should be the same. The preferred host species (*Distichlis spicata, Monanthochloe littoralis*, and *Salicornia virginica)* should be well established on the site, with canopies that are not too open or too dense (60–80% shade). Habitat for ground-nesting bees should be provided immediately adjacent to the transplant site. The mechanisms that maintain open canopies of host plants should be present. Small mammals are most likely responsible for the openings, through both burrowing and browsing. Until determined otherwise, it is recommended that habitat for small mammals be provided adjacent to the transplant site, although animals would not need to be introduced until canopy closure becomes a problem. Finally, it is recommended that many well-separated sites be available for growing bird's beak, so that disturbances that might wipe out one colony would not occur throughout the transplanting location.



The mitigation agreement should require a set number of colonies of bird's beak. Specific performance standards would then be based on the size of each colony (total plant counts), an estimate of seed production (mean number of seed capsules set per plant), and further observations of limiting factors at each transplanting site. Three to five colonies at Tijuana Estuary could be used as references, and when the numbers of plants produced are at least 90% of the mean colony size, and numbers of seed capsules produced are statistically indistinguishable from those at Tijuana Estuary for perhaps 5 consecutive years, the site could be judged in compliance. It is possible that some colonies will fail during the project, as happens in natural marshes; hence, mitigators would be wise to establish and to monitor extra colonies in order to show that the required number has persisted for the requisite time.

Performance standards for mid- and high-elevation marsh vegetation. The ability of salt marshes to support native plants and animals depends on the degree of tidal flushing (PERL 1990). Tijuana Estuary, with a long history of tidal flushing, supports 19 native halophytes, whereas as few as three occur in sites with little saltwater influence. Expectations for fully tidal marshes should thus be greater than those for sites with muted tidal flow, that is, tidal action provided through culverts or over weirs that may be needed to prevent flooding of adjacent real estate. The three halophytes that occur in all Southern California coastal wetlands are *Distichlis spicata, Frankenia grandifolia,* and *Salicornia virginica.* These same species are also often dense and widespread in their distribution. At least the last two are highly variable in relation to changes in the degree of tidal flushing (cf. Ferren's [1985] historic aerial photos of Carpinteria Marsh and information on pickleweed dynamics at Tijuana Estuary [Zedler, Nordby, and Kus 1992]). Species of broad ecological tolerance are more likely to do well in restoration sites than species with narrow tolerance ranges for moisture, salinity, and inundation, so general standards may suffice for the former. Dominance by one or more of these three halophytes, equivalent to that of the reference marsh(es), may be a suitable performance standard.

Pickleweed is a "matrix" species of the mid-marsh plain, and Belding's Savannah sparrows use the tallest stems as perches to defend territory and to attract mates (White 1986; Powell 1993). Hence, standards for canopy cover and maximum height (not mean height) are needed. Sampling reference marshes by using transect lines (intercept within 10-cm intervals) and determining maximum height in each 1-m interval is suggested for establishing criteria for suitable sparrow nesting habitat.

The matrix species of the high marsh are *Salicornia subterminalis* (glasswort) and *M. littoralis.* Of these, tall glasswort has at least three important roles in the high marsh: It provides cover for clapper rails that move out of the low marsh during high tides; it serves as a perch for Savannah sparrows; and it supports a myriad of spider webs, again above the high tide line. Canopy structure should thus be measured and maximum heights should be required to be similar to those in the reference wetlands, and the abundance of tall plants in the two wetlands should be comparable. It is recommended that height histograms be developed for constructed and reference sites and that similar frequency of the tallest stems be required.

Species that are rarer and more patchy need more attention and different standards to ensure a diverse marsh community. Frequencies of occurrence should be assessed more broadly (over larger areas than readily sampled by using transect lines) and with less attention paid to their overall cover. Presence in 1-m intervals or in 0.25-m^2 circular quadrats should suffice for their occurrence. On the basis of our current understanding of the functioning of nondominants, an appropriate goal is their continued presence, rather than some specific height, cover, or biomass.

Species that are deemed sensitive, that is, threatened or endangered with extinction, require the greatest attention. At present, these include bird's beak, Coulter goldfields (*Lasthenia glabrata),* and others listed by Ferren (1985) and PERL (1990).

Total cover (=100% minus bare space) is an additional attribute that requires performance standards. It is reasonable to require minimum standards for the percentage of a site that needs to be vegetated, that is, the maximum amount of open space allowed. Natural marshes typically have quite high cover, with few open patches in the canopy. Previous sampling programs have not measured total cover, because the tendency in monitoring vegetation has been to place transects in homogeneous areas of continuous cover. Tide pools and gaps caused by disturbances have typically been avoided. Because open space is patchy in natural marshes, on-the-ground sampling is likely to underestimate or overestimate it, depending on the bias of the sampling technique. A new method for determining bare space uses airborne digital acquisition and registration imagery from low-elevation flights (Phinn and Stow, Section 6.4). A total-site summary of bare space can easily be made with resolution of less than 1 m^2. In addition, the size of each patch, shape (perimeter and area), and distribution of open patches can be assessed. Thus, remote sensing is the recommended procedure for assessing total cover of restoration and mitigation sites.

Finally, there should be a performance standard for the maximum allowable cover of exotic plants. Airborne digital acquisition and registration imagery is also proving valuable in mapping and summarizing the distributions of the more conspicuous invasive exotic plants, such as pampas grass (*Cortaderia selloana),* giant reed (*Arundo donax),* and Hottentot fig (*Carpobrotus edulis).* By definition, undisturbed wetlands would have no exotic species (plant introductions to the region are a disturbance), and reference wetlands will differ in the degree to which they have been disturbed.

A zero-tolerance policy may be best for the most aggressive invaders: pampas grass, giant reed, Hottentot fig, rabbitfoot grass *(Polypogon monspeliensis),* and curly dock *(Rumex crispus).* The presence of exotic vascular plants can and should be avoided by adjusting environmental conditions to preclude their establishment or spread. Addition of salt is proving to be a useful control measure (Kuhn 1995). Exotic species that become established should be eradicated to prevent spreading. Even those that are naturalized in many natural marshes (e.g., sickle grass *[Parapholis incurva],* brass buttons *[Cotula coronopifolia],* and ice plant *[Mesembryanthemum crystallinum* and *M. nodiflora])* need to be minimized in constructed wetlands. Recommendations for additions of salt (timing, amount, and duration) that can eliminate exotic species yet allow native annuals to germinate, grow, and reproduce are being developed.

4.2 A REGIONAL APPROACH TO PLANNING*

Mitigation plans often include off-site projects, out-of-kind mitigation, and mitigation banks. Plans that provide mitigation credit—not for creating new wetland habitat but for converting one type of wetland to another—result in a net loss of area, because both the affected site and the mitigation site are modified, but only the latter is "repaired." This is a swap of quantity for quality, and it seems to violate the no-net-loss policy. Regulatory agencies agree, however, that the minimum replacement ratio "may be less than 1 to 1 for areas where the functional values associated with the area being impacted are demonstrably low and the likelihood of success associated with the mitigation proposal is high" (Environmental Protection Agency and U.S. Army Corps of Engineers 1990, pp. 5–6).

The restoration of tidal action to Batiquitos Lagoon (City of Carlsbad and U.S. Army Corps of Engineers 1990) is an off-site, out-of-kind mitigation project. Nearly the entire lagoon is being dredged to mitigate damages to deepwater habitat that will be lost when the Port of Los Angeles (145 km north of the site) fills part of San Pedro Bay to improve port facilities. Subtidal areas are being excavated to support fishes and invertebrates; the tidal mudflats would support shorebirds, and sandy island sites (added in 1994) are to support least tern nesting activities. Reintroduction of tidal flows should benefit the salt marsh remnant that persists at the eastern end of the lagoon. The ecological issues concern the extent of dredging needed to accomplish these benefits, the ecological costs of eliminating the shallow, nontidal lagoon, and the effect of extensive and long-term dredging.

Although remaining wetlands are disturbed, they still support sensitive species. For this reason, further disturbances carry a risk, even when the modifications are designed to enhance the sites. The effects of dredging over long periods are unknown. If large areas of little-occupied habitat were available elsewhere in the region, birds and other animals might be able to move away during construction. Absence of such areas may mean heavy mortality during construction. If large reserves of animals were present elsewhere, perhaps regional biodiversity could better withstand temporary dredging of coastal lagoons.

Because the fish species that use deepwater harbors are not likely to be the same as those found in coastal embayments, this mitigation project is somewhat out-of-kind. Perhaps even greater latitude should be taken in mitigation trade-offs. Within the regional context, the most important goal is sustaining biodiversity. What is most needed is a plan that will avoid further adverse effects or, when these are truly unavoidable, allow use of mitigation funds to accomplish regionwide goals in a sequence that will most enhance populations of native species and offer the species' respective ecosystems a greater potential for sustainability (Zedler 1996b).

*Modified from Zedler 1996b. ©Ecological Society of America. Printed with permission.

4.2.1 Regional Losses

Little quantitative information is available on the historical condition or functioning of Southern California's two dozen coastal wetlands. For three tidally flushed sites, the historical changes in habitat type were inventoried by Macdonald (1990; Table 1.2). The shallow-water habitats (salt marsh and intertidal flats) of these sites have declined to 15% of the habitats' historic area, whereas subtidal area has remained stable (although most deepwater sites have been dredged). Most of the other wetlands are closed or often closed lagoons (Zedler 1982) that were historically more influenced by ocean waters. Construction of roadways through wetlands, together with increased sedimentation from development in the watersheds, has reduced tidal prisms and facilitated formation of berms across their inlets. Tidal influence has declined, and salinity regimes have become more extreme as a result, with the impounded waters becoming mildly brackish when freshwater inflows are dominant or extremely hypersaline when evaporation predominates (Carpelan 1967). The effects of these changes on wetland structure or functioning have not been quantified.

Managers and scientists agree that increased tidal flushing is desirable for support of more sensitive species (Section 2.2), and several enhancement and mitigation projects are planned to accomplish this objective (Table 4.1). The projects at Anaheim Bay, Batiquitos Lagoon, and San Dieguito Lagoon are expressly designed to enhance fish habitat as mitigation for loss of deepwater fish resources. Others (e.g., the 200-ha project proposed for Tijuana Estuary) would attempt to restore what has been lost at that wetland in recent decades. If all the planned projects are eventually implemented, significant, major changes will be made to the region's coastal landscape with little understanding of the consequences, unless we discover what functions are unique to the shallow lagoons

With so many projects in the region's future, it makes sense for the final design and implementation phases to be coordinated, so that the ultimate landscape has an optimal mix of habitat types, however defined. Projects should not proceed independently nor be driven solely by the needs of the mitigator. It is not uncommon for an individual project to include a standard list of targets (Table 4.2) with little regard for the potential of the mitigation site to sustain all functions. The continuing needs of port districts to find sites to compensate for loss of deepwater fish habitat portend continued conversion of shallow-water wetlands to deepwater channels and lagoons. The best quantitative

Table 4.1. Restoration and Enhancement Projects Planned for Southern California Wetlands*

Site	Proposed Change
Goleta Slough	Remove obstructions to increase tidal flows
Carpinteria Marsh	Excavate adjacent site to increase area of tidal influence
Ballona Wetland	Open tide gates and increase tidal flows to salt marsh
Los Cerritos Wetland	Create access to tidal flow in diked marshes
Anaheim Bay	Maintain tidal system (tidal basins for fish have been constructed to mitigate habitat loss in San Pedro Bay)
Bolsa Chica Wetland	Enhance tidal access to diked marshes and flats
Santa Margarita Estuary	Dredge mouth to extend periods of tidal influence
Agua Hedionda Lagoon	Dredge mouth periodically to sustain tidal influence
Batiquitos Lagoon	Dredge lagoon to shift it from nontidal to tidal
San Elijo Lagoon	Dredge mouth to shift it from nontidal to tidal
Los Peñasquitos Lagoon	Dredge mouth periodically to extend periods of tidal flow
San Dieguito Lagoon	Shift lagoon from nontidal to tidal
Famosa Slough	Enlarge culverts and gates to increase tidal flow rates
Mission Bay	Remove fill to restore salt marsh
Sweetwater Marsh on San Diego Bay	Excavate fill to create intertidal marsh (12 ha have already been excavated)
Tijuana Estuary	Dredge upland and transitional areas to create salt marsh and tidal-creek systems

*From Zedler 1996b

data available on habitat loss (Table 1.2) suggest that the greatest need is to restore salt marshes and intertidal flats, rather than deepwater habitats.

We do not know exactly how the current wetlands developed, i.e., what processes shaped existing communities or what rare and extreme events influenced the occurrence and abundance of species. We do not know the dependencies among the components of the wetland; competitive and predator-prey interactions are poorly understood. We cannot predict the dynamics of various populations or that they will persist in perpetuity. We do not know what factors confer resilience to species and communities.

4.2.2 The Need for a Regional Restoration Plan

Two alternatives are suggested to sustain wetland biodiversity. The first is to halt damages to wetlands. In mitigation policy, the first priority is avoidance of adverse effects. Unfortunately, it is not easy for the regulatory agencies to deny permits for activities on private land. Even on public land, developments are possible if the project is deemed "good" for the public. Thus, in the San Diego area, Interstate Freeway 5 has been widened by filling parts of several coastal wetlands; access roads have encroached on the periphery of Agua Hedionda Lagoon, Batiquitos Lagoon, and Los Peñasquitos Lagoon; a flood control channel dissected a wetland complex along San Diego Bay; a sewer pump station is being relocated and expanded within Los Peñasquitos Lagoon; and trail systems are proposed to increase public access into Tijuana Estuary and Sweetwater Marsh.

The second alternative is to conduct all enhancement projects within a regional wetland restoration plan, a coordinated effort that encompasses 320 km of coastline and includes the following elements:

- Characterize the existing resource base (existing area of each habitat type, patch size and shape, connectivity, isolation) by using National Wetland Inventory maps and remote sensing imagery.
- Determine the unique qualities of each habitat type (e.g., impounded lagoon waters may support species that cannot use tidal waters).
- Set regional goals for sustaining biodiversity, and establish general plans for retaining and increasing each habitat type.
- Use existing plans (Table 4.1) and suggestions to identify and map all potential restoration and enhancement sites.
- Indicate the most appropriate restoration procedures for each site. Several ecological principles can guide these decisions (Table 4.3) within constraints on what can be accomplished (e.g., Ballona Wetland and Famosa Slough require tide gates to protect urban areas from inundation, which precludes a goal of restoring full tidal flow).
- List existing mitigation needs, that is, projects that have been proposed.
- Match needs with opportunities insofar as is possible, favoring in-kind mitigation.
- Where no match is possible or where extensive modifications are unlikely to be successful, establish out-of-kind compensations that contribute to regional goals.
- Throughout the process, follow an adaptive management approach and carry out research to understand successes and failures.

A suitable regional priority would be reestablishment of habitats that have been most reduced in area or species. For Southern California, the following steps could be taken:

Table 4.2. Habitat and Species Commonly Listed as Objectives or Targets in Coastal Wetland Restoration Projects in Southern California

Habitat Type	Sensitive Species
Tidal cordgrass marsh	Light-footed clapper rail (*Rallus longirostris levipes)*
Pickleweed marsh	Belding's Savannah sparrow (*Passerculus sandwichensis beldingi)*
High intertidal marsh	Salt marsh bird's beak *(Cordylanthus maritimus* spp. *maritimus)*
	Wandering skipper (Lepidoptera: *Panoquina errans)*
Salt pan	Tiger beetle (Coleoptera: *Cicindela* spp.)
Mud flats	Shorebirds
Saline tidal channels	California halibut (*Paralichthys. californicus)*
Riparian willows	Least Bell's vireo (*Vireo bellii pusillus)*
Bird nesting islands	California least tern (*Sterna antillarum browni*)
	Western snowy plover *(Charadrius alexandrinus nivosus)*

- First, determine how each site functions as a biodiversity reserve.
- Expand the area of these reserves where possible, emphasizing habitat types that have most declined in area. This will improve chances for attracting species and sustaining the full complement of native plants and animals. Tijuana Estuary represents such an opportunity in Southern California. It supports 24 sensitive species, and a 200-ha restoration plan has been developed (Entrix et al. 1991), yet there are no funds to implement it.
- Next, expand the area of small wetlands, emphasizing the type of habitat that is still present. For example, at San Dieguito Lagoon, existing vegetation *(Salicornia virginica* and associated species) could be expanded more easily than introducing cordgrass *(Spartina foliosa).*
- Once the expanded biodiversity reserves show promise for attracting and supporting sensitive species, create such ecosystem types at wetlands that formerly included them. For example, Los Peñasquitos Lagoon had lush cordgrass in 1939 (Purer 1942), although it has disappeared, along with clapper rails.
- The last priority would be to create such habitats at sites where they may never have occurred historically. There is no historical record of cordgrass at San Dieguito Lagoon, for example.

The ongoing planning for enhancement of San Dieguito Lagoon has incorporated several of these elements. The California Coastal Commission is requiring Southern California Edison to restore 60.75 ha of self-sustaining functional wetland, with a combination of in-kind (substantial fish habitat) and out-of-kind (provision for rare species) objectives, as a condition for continued operation of the San Onofre Nuclear Generating Station. Eight wetlands were initially selected as potential mitigation sites for Southern California Edison. Of these, San Dieguito Lagoon and Tijuana Estuary ranked highest (MEC 1991), and the former was selected after a lengthy review and public hearing.

Other recommendations included restoration of about 36 ha at San Dieguito Lagoon, where channels might be readily enlarged to accommodate fish habitat, and about

Table 4.3. Scientific Principles That Can Help Guide Choices for Wetland Restoration*

1. Large systems support and maintain the highest biodiversity. Thus, restoration work with large systems should have greater potential for sustaining regional biodiversity.
2. Coastal wetlands support greater biodiversity if there are good linkages with adjacent ecosystems (uplands, riparian corridors, and nearshore waters) and few barriers to water flow and movement of animals (e.g., culverts, roads, light and telephone wires). Adjacent habitats provide high-tide refuges (resting places and alternative foraging areas for animals that avoid tidal inundation), habitat for upland species that are essential to wetland species (e.g., insect pollinators that live in uplands but are needed by annual wetland plants), access to water-borne larvae and plant propagules (for establishment and reestablishment of native species), and areas for wetland vegetation to "migrate" as the sea level rises (essential for long-term maintenance of biodiversity). Thus, restoration projects should remove barriers and improve connectivity.
3. Specific ecosystem types will develop best if located near or adjacent to an existing ecosystem of the same type. Some desired species can be transplanted (e.g., plants), but the rest of the native community must invade and become established on its own. Nearby sources should offer higher probability for dispersal.
4. Small habitat remnants are likely to have reduced resilience and less resistance to natural and man-made perturbations (i.e., fewer refuges from disturbance, more exotic species, fewer propagules, slow recovery rates). It may be more important to retain and expand on the habitat remnants at the smaller wetlands, rather than add new ecosystem types.

*From Zedler 1996b.

24 ha at Tijuana Estuary, where the chances of augmenting populations of endangered species are high. However, Southern California Edison argued successfully for a single site. Initial hydrologic studies have since shown that full tidal flushing is hard to achieve at San Dieguito Lagoon and that frequent maintenance dredging will be essential. Because the mitigator must maintain tidal flow only so long as the San Onofre station is operated, rather than in perpetuity, it would be less risky to construct habitats that can withstand tidal closure (e.g., the pickleweed community, which supports Belding's Savannah sparrows), rather than to count on long-term tidal flushing.

For long-term maintenance of regional biodiversity, each ecosystem type should consist of larger areas and be present in more locations, as was the case historically. This is particularly difficult to achieve along the coast of Southern California, where little open space remains. This region may represent a worst-case scenario for mitigation problems; it certainly challenges mitigators, resource agencies, decisionmakers, and researchers. In regions with a larger proportion of the natural habitat remaining, the potential for on-site, in-kind mitigation should be greater. Still, a regional framework can only improve chances of fulfilling the nation's no-net-loss policy, that is, sustaining wetland area and function.

4.3 THE TIJUANA ESTUARY TIDAL RESTORATION PLAN

The Tijuana Estuary Tidal Restoration Plan is a model restoration project. First, it would restore what has been lost at the site (approximately 150 ha of salt marsh and 48 ha of intertidal flats) rather than being driven by the need for mitigation of other types of habitat loss (e.g., nearshore subtidal fish habitat; see Humboldt Bay case study in chapter 2). Second, it follows an adaptive management program, recognizing that efforts to restore habitat, particularly for endangered species, are not yet assured of success. In order to help develop needed techniques, restoration would be done in phases. The first phase includes an experimental 8-ha marsh that will have areas with and without extensive tidal networks, allowing an ecosystem-level assessment of the need for intricate topographic sculpturing. The plan further requires experimentation with salvaging of vegetation and soils when damage is unavoidable. The remaining part of the 198-ha restoration plan would be implemented in modules, with lessons learned from earlier modules incorporated into plans for later modules.

Tijuana River National Estuarine Research Reserve includes the Tijuana Slough National Wildlife Refuge, managed by the U.S. Fish and Wildlife Service, and Border Field State Park, managed by the California Department of Parks and Recreation. Its specific goal is to protect three endangered species: the light-footed clapper rail, California least tern, and salt marsh bird's beak.

A program of adaptive management (Walters and Hilborn 1978) with long-term monitoring, field studies, manipulative experiments, and a major restoration program guides resource management at Tijuana Estuary (Entrix et al. 1991, Zedler et al. 1992).

4.3.1 Management Needs

Tijuana Estuary has many management needs. It is an urban estuary subject to the long- and short-term effects of a large human population (cf. section 3.3). The city of Imperial Beach surrounds the northern arm of the estuary. Flood control levees in agricultural lands have modified the Tijuana River floodplain. Just upstream, the city of Tijuana, Mexico, has about 2 million inhabitants. Sewer hook-ups are few, and human wastes flow north into the Tijuana Estuary. The watershed above Tijuana is scheduled for rapid and extensive development. At the same time, the downstream estuary is supposed to serve as a reserve for research and education, a refuge for endangered species, and a state park.

Sedimentation problems. Increased sedimentation follows disturbance of soil-stabilizing vegetation within the watershed. Throughout Southern California, estuaries and lagoons have been filling in rapidly, as hillsides within their watersheds are disturbed and developed. Vegetation that might slow erosion is sparse in Mexico, where grazing is more common and fires occur more often than in the United States. The extremely erodible soils move downstream with winter rains, and catastrophic sedimentation occurs at the coast. Mugu Lagoon lost 40% of its low-tide volume because of cumulative sedimentation during the floods of 1978 and 1980 (Onuf 1987). Sedimentation is a natural process, but the rates have accelerated, and the lagoon's ability to respond by changing its configuration has been constrained by peripheral developments.

Natural events that counter the effects of sedimentation may also be augmented by human activities. For example, the greenhouse effect may be accelerating the rise in sea level and compensating somewhat for sediment accretion. However, the average rise in sea level cannot begin to keep up with catastrophic sedimentation accompanying major floods; the net effect is that coastal wetlands fill in more rapidly than they would naturally.

Problems of beach and dune erosion. The beach and dunes erode when winter storms and high sea levels coincide. Dunes contribute sediment that fills channels in the adjacent estuary. Summer is the rebuilding phase in an annual cycle of beach removal and replenishment. However, if the replenishing sands are intercepted in their transport along shore or downstream, the beach and dunes show a net loss (Inman 1985). In addition to the annual cycle, the long-term trend is an increase in mean sea level, which gradually moves the beach inland.

Disturbance of beach vegetation has contributed to the destabilization of the dunes, and replanting efforts have been under way in recent years (P. Jorgensen and

B. Fink, unpublished data). Further dune stabilization, through extensive revegetation efforts, is called for in the restoration plan.

Problems in habitat management. For an estuary that is managed primarily for its native wetland communities and endangered species, passive management (leaving nature alone) might seem preferable to active manipulation of environmental conditions. However, disturbances have had significant effects on the estuary, and the issue is not whether, but how much, intervention is required to maintain native species.

Decades of disturbance to the estuary and its watershed have substantially altered the environmental factors that control habitats. Since 1900, some communities have been lost entirely (e.g., woody beach vegetation), and other new ones have developed (e.g., brackish ponds and marshes). The overall management goal is to maintain the natural variety of habitats, recognizing that increasing the area of any one habitat type should not reduce habitat for another. Single-species management is not desirable, because procedures that might benefit one species might adversely affect another. Restoration requires a comprehensive approach, because altering habitat in one area can potentially affect the entire ecosystem. Habitats that need to be restored include the following:

- Transition from upland to wetland
- Salt marsh
- Intertidal salt marsh
- Salt pans
- Channels and creeks
- Sand flats and mud flats
- Beach and dunes
- Riparian ecosystem
- Coastal sage scrub

4.3.2 The Tidal Restoration Plan

The goals for Tijuana Estuary are to improve tidal flushing enough to maintain an open ocean inlet and to create and restore sufficient habitat for maintenance of estuarine biodiversity. These goals are complementary; excavating sediments that have washed into the estuary will increase tidal flushing and expand wetland habitats. Most of the restoration work will be done in the southern arm of the estuary, where sedimentation has been heaviest in the past and where tidal flows are most sluggish.

In 1984, the State Resources Agency and the State Coastal Conservancy provided funding to map the Tijuana River National Estuarine Research Reserve and to develop a hydrologic model (Williams and Swanson 1987). This was followed by a 3-year assessment of resources and evaluation of environmental impact funded by the State Coastal Conservancy in 1988. Plans to restore full tidal flushing to the southern arm of Tijuana Estuary began with the hydrologic analysis. Williams and Swanson (1987) used the 1857 map as a model of what the estuary might need to become a fully tidal estuary. In 1857, the mouth was 305 m wide, and the tidal prism was estimated to be 1.8 million m^3. About 352 ha of intertidal wetlands were estimated to be present. Tidal sloughs extended 914 m inland toward the east, 1524 m north, and 610 m south from the ocean inlet. A large area of open water was present in the southwesternmost part of the estuary.

Between 1852 and 1986, many changes occurred in geomorphology, hydrology, and hydrodynamics (Fig. 4.2). From the historic maps and aerial photos, it was estimated that 80% of the historic tidal prism had been eliminated. Sediments flowed in from the watershed and across the beach, depending on the type of storm event. With sea storms, the dunes were both flattened and pushed inland. Major retreat of the beach (90–120 m) was documented for the 134-year period.

Williams and Swanson (1987) summarized the major problems that would continue to plague the estuary if these physical processes were not abated: (1) The estuary would continue to lose its tidal prism, through sedimentation down Tijuana River, down Goat Canyon, and across the beach during storm washovers. (2) Wetlands would continue to be converted to upland habitat. (3) Freshwater inflows could shift large areas of saline wetland to brackish wetland. (4) Wetland area would continue to decline as a consequence of accelerated rise in sea level. (5) The encroachment of development in the river corridor would degrade the riparian habitat and increase flood hazards.

The hydrologists suggested an extensive dredging program (Fig. 4.3) that would focus on the southern arm of the estuary. They also proposed a major new channel in the northern arm to connect the tidal ponds to the Tijuana River channel, a measure that would maintain tidal access to Oneonta Slough after the migrating dune pinches off its current opening to Tijuana River and the ocean inlet.

Mapping of resources and assessment of environmental impact followed, with preparation of an environmental impact review and an environmental impact statement by Entrix et al. (1991). A biological analysis was undertaken to determine what resources would be affected by the proposed dredging (Enrix et al. 1991).

A constraints map (Fig. 4.4) was developed from the field studies; it details all areas considered sensitive, either for the type of community present or the occurrence of rare and valued species. For example, the area of transition to upland that occurs south of the tidal ponds was considered too valuable to be dredged. Salt flats and high marsh habitat near the Border Field overlook had high densities of Belding's Savannah sparrows and hence were considered too sensitive for conversion to low marsh.

Modifications to the 1987 plan were suggested to incorporate new information and ideas. It was proposed that a channel be cut near the visitor center at Tijuana

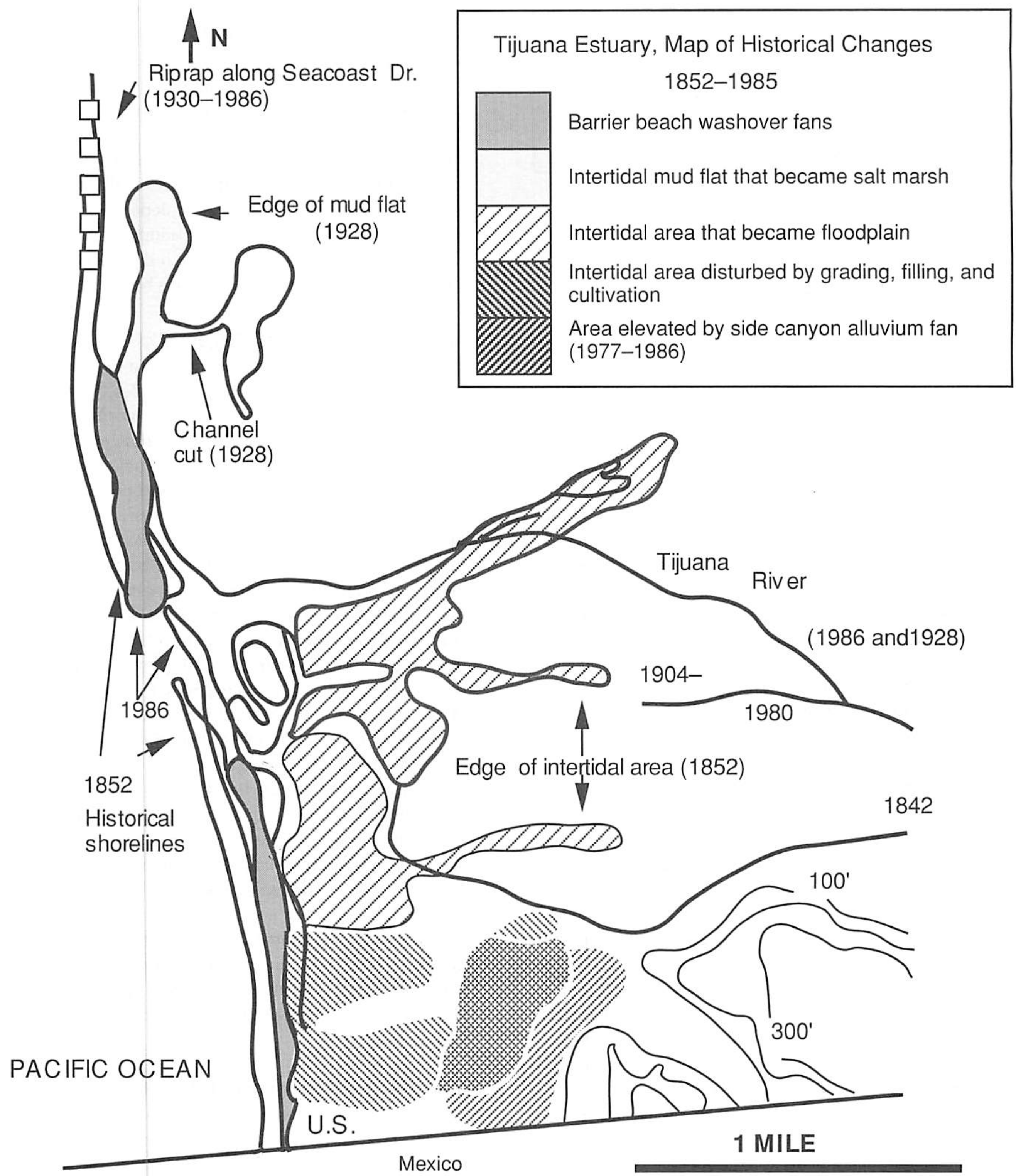


Figure 4.2 Historical changes that have reduced the tidal prism of Tijuana Estuary. Modified from Williams and Swanson 1987.

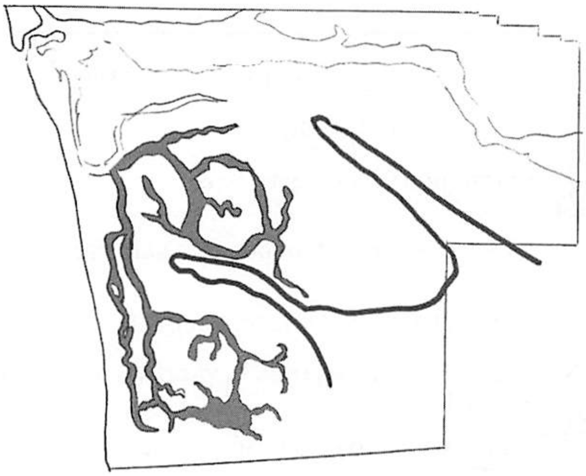


Figure 4.3 Tijuana Estuary "minimum dredging plan" designed to restore the tidal prism. Two river training berms (bold lines) were designed to divert sediment inflows away from excavated channels and to accommodate dredge spoils. Modified from Williams and Swanson 1987.

Estuary to improve tidal circulation, enhance opportunities for nature interpretation, and reduce the need to bring visitors into habitats of endangered species. The hydrologists were asked to redesign the dredging program to avoid sensitive areas. Florsheim et al. (in Entrix et al. 1991) redesigned the hydrology plan and responded to recommendations for an adaptive management approach, with two phases (a model project to precede full-scale restoration) and a modular approach (subunits that could be constructed as funding became available).

4.3.3 The Model Project

The model project has three features: First, an experimental marsh (Fig. 4.5) of at least 8 ha will be constructed to determine how excavation of tidal creeks will accelerate development of the ecosystem (cf. section 4.3.5).

The second feature is a new channel to connect the northern end of Oneonta Slough to the northern tidal pond. In addition to improving tidal flows, it will enhance nature interpretation and increase habitat acreage. The channel will be excavated along the toe of the fill on which the visitor center is located and will provide ready access for interpretive activities. Several of the estuary's habitats will be created within the excavation, including tidal creek, mud flat, lower-to-upper marsh, and transition to upland. It is likely that the public will join the project, much as volunteers have participated in planting and weeding the coastal scrub vegetation that landscapes the site of the visitor center.

The third feature is the widening of Oneonta Slough where a buried hard pan prevents inland migration of the channel. Unless the hard substrates are removed, tidal flows would eventually be pinched off as the dune gradually encroaches from the west. Maintaining Oneonta Slough as the tidal access for the northern arm of the estuary will make it unnecessary to cut a new channel south of the tidal ponds, thus preserving the transitional wetland-upland habitats that occur there.

Disposal of dredge spoils remains an issue. Dredging will generate spoils of various qualities. Sandy spoils may be used to replenish the beach and dunes, but fine-textured spoils will need to be taken off site. High salinity and presence of contaminants will dictate the spoils' ultimate disposal. Off-shore deposition in an authorized deep-sea dump site is an additional option. At this writing, only the plan for beach disposal of spoils from the second and third feature has been approved.

4.3.4 The 200-Hectare Project

It is expected that the 200-ha excavation (Fig. 4.6) will be implemented over two or more decades, because the project will be costly and funds are not currently available for the work. An innovative, modular approach will accomplish two adaptive management objectives. First, it will be possible to match each funding opportunity with the restoration of one or more habitat modules. Rather than proceeding piecemeal, the restoration will be completed in modules to make up the 200-ha program. Second, monitoring of and research on each completed module will improve the next. As problems are recognized, corrective measures can be built into later construction plans. As restoration methods are improved or new ideas developed, they can be incorporated into subsequent modules. The essence of adaptive management is a dynamic plan that can improve as knowledge accumulates and restoration science progresses.

Protection from future flooding and sedimentation is not assured. Because Tijuana Estuary has accumulated vast amounts of sediment in recent floods, there is always the possibility that the restoration site will be refilled by flood-borne sediments. It is not yet clear how this can be avoided or accommodated (cf. section 3.1). The original restoration plan called for two river training berms (built from dredge spoils); the larger would be 2.4 km long and 18 m high. Subsequent plans eliminated both berms because the environmental impacts were considered excessive.

4.3.5 Restoration Research Needs

Several questions remain about how to conduct the restoration project. In response to these needs, the restoration plan (Entrix et al. 1991) listed several experimental projects, pointed out alternative restoration measures, and called for supplemental analysis of the environmental impact once choices are made. Issues requiring further research and analysis include the following:

Efforts to salvage soil and vegetation. Dredging to restore channels will disrupt small areas of benthic animal communities and creekside vegetation and infauna. To the extent possible, areas of native vegetation will be avoided; where unavoidable, a salvage and revegetation

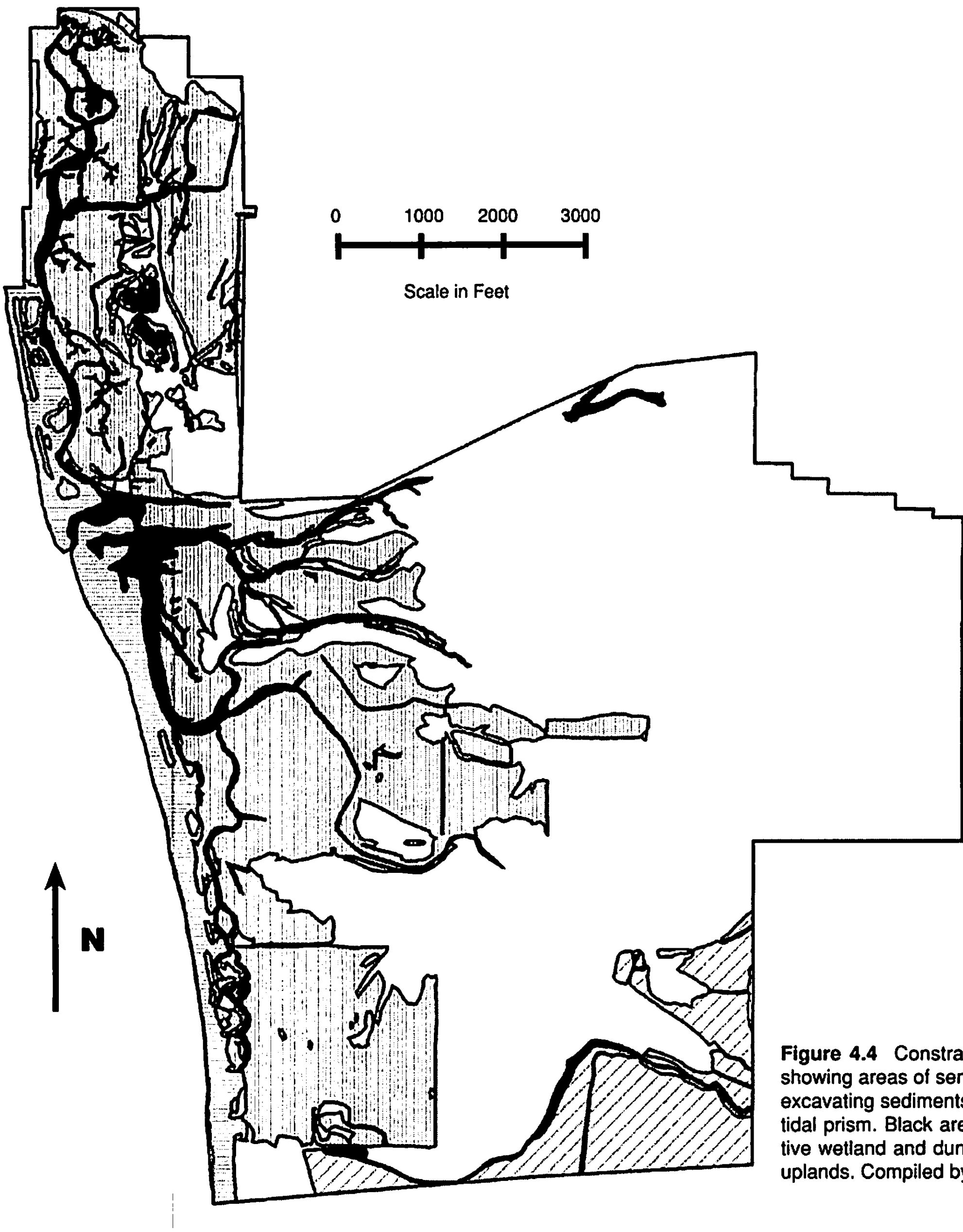


Figure 4.4 Constraints map for Tijuana Estuary showing areas of sensitive habitat to be avoided in excavating sediments for purposes of restoring the tidal prism. Black areas = water; shading = sensitive wetland and dune areas; hatching = sensitive uplands. Compiled by PERL.

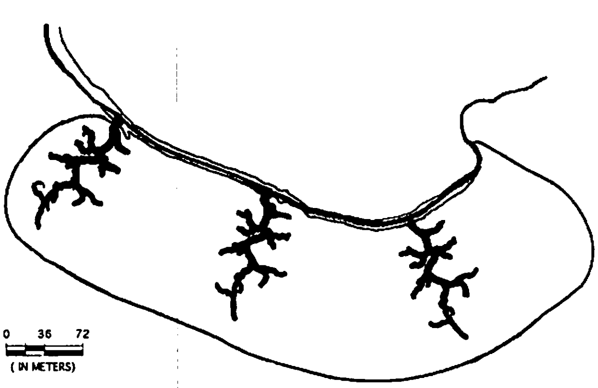


Figure 4.5 The 8-ha model marsh that has been planned to allow experimental comparison of areas with and without tidal creek networks.

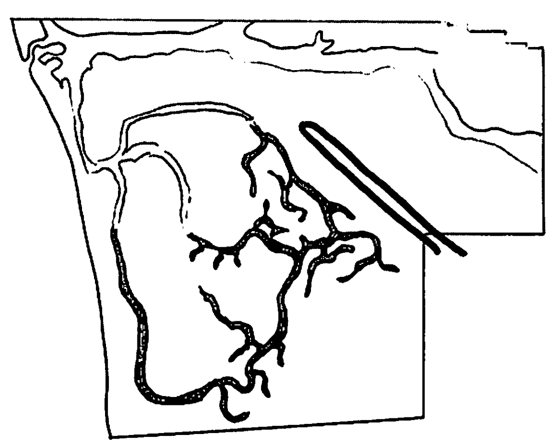


Figure 4.6 Revised tidal restoration plan for Tijuana Estuary. The river berm (in bold) was later eliminated. Modified from Entrix et al. 1991.

program will be undertaken. As discussed later (cf. section 5.1.1), the fine sediments will be needed to provide a suitable substrate for the proposed tidal marsh. There are no precedents for sediment salvage work of this type, and details have not been planned. Experiments are being developed to determine the size and depth of soil cores needed to retain native vegetation and soils, including root zones and seed banks.

Propagation and replanting. When the first restoration site has been excavated, studies will compare rates at which species become established with and without revegetation. The new channel adjacent to the visitor center will most likely house these experiments. There will be many opportunities for public interpretation, making Tijuana Estuary a major demonstration site for state-of-the-art restoration methods.

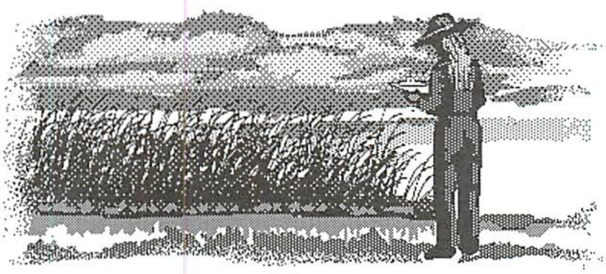
Ecosystem development with and without tidal-creek networks. Several functions of wetlands are hypothesized to relate to the presence of tidal creeks within the salt marsh plain. These are rapid increase in nutrient pools, enhanced plant growth, denser and more diverse invertebrate populations, and enhanced use by birds. To test these dependencies, the 8-ha marsh will be subdivided into six areas, three with and three without tidal-creek networks incised into the marsh plain (Fig. 4.5). Development rates of the ecosystem under these two treatments will be compared to determine the need for construction of tidal creeks, which will add to the initial cost of marsh construction throughout the 200-ha restoration program but reduce the time required for achieving the goals of restoration.

CHAPTER 5

Methods of Enhancing Ecosystem Development

5.1 AMENDING SOILS TO PROVIDE TALL CORDGRASS CANOPIES

Cordgrass canopies fail to grow tall and robust on sandy, nutrient-poor sediments. Using agricultural principles, it follows that sediments could be amended in ways that would build up the soil, improve the nutrient status, and increase plant height. Yet developing specific recommendations has proven complicated.



5.1.1 The Importance of Organic Matter

Organic content is one of three key factors (with elevation and drainage) that influence marsh productivity (Mitsch and Gosselink 1986). In mature wetland systems, the organic matter in sediments is the major storage pool of nutrients (Haines 1977, Langis et al. 1991), and a large part of these nutrients is stored and recycled within the ecosystem (Mitsch and Gosselink 1986). The importance of organic matter for nitrogen cycling is not sufficiently clear, but we do know that nitrogen is a limiting nutrient in salt marshes (Valiela et al. 1976, Covin and Zedler 1988). Both nitrogen fixation and denitrification rely on the type and amount of organic matter present. Additions of organic matter stimulated nitrogen fixation in both natural and constructed marsh soils along San Diego Bay (Zalejko 1989). Organic matter in sediments also improves the capacity for cation exchange.

Unfortunately, low levels of organic matter in sediments seem to be characteristic of constructed wetlands along the coast, not only in Southern California (Langis et al. 1991) but also in North Carolina (Craft et al. 1986, 1988) and Texas (Lindau and Hossner 1981) (cf. Table 3.1). These studies agree that low levels of organic matter limit nutrient supply and slow the rate of functional development in constructed marshes.

Organic matter accumulates slowly in marsh sediments. Accumulation occurs when decomposition rates are low, because of low oxygen levels and inhibitory acids in anaerobic soils at low pH (Kilham and Alexander 1984). The major sources of organic matter in the soil in salt marshes are thought to be emergent macrophytes (Craft et al. 1988), especially those with dense root systems and particulate matter that settles out of the water column. For the Pacific Coast, accumulation of peat is rare to absent (Frey and Basan 1978). The natural reference marsh at San Diego Bay (Paradise Creek) had only 2.0–2.5% organic carbon (percentage dry mass; Langis et al. 1991). Because it is uncertain how long a constructed marsh will take to match the organic levels and nutrient status of natural wetlands, the time required for functional equivalency cannot be predicted.

5.1.2 Shortcomings of Sandy Substrates at San Diego Bay Mitigation Sites

Study of a 5-year-old constructed coastal wetland in San Diego Bay indicated that sandy dredge spoils are poor substrates for intertidal wetlands. The sediments had low levels of organic matter and nitrogen, which impaired the development of ecological functions (Langis et al. 1991). Soils and vegetation at the constructed Connector Marsh and the natural Paradise Creek Marsh were compared in detail when the former site was 4–5 years old. Significant differences were found in soil organic matter, nitrogen, nitrogen-fixation rates (Zalejko 1989), sulfate reduction, and sulfide production (Cantilli 1989). Low concentrations of nitrogen appeared to limit both plant growth (Langis et al. 1991) and consumer species. Epibenthic invertebrates were only a third as abundant at Connector Marsh as at Paradise Creek Marsh (Rutherford 1989; Scatolini and Zedler 1996).

In the summer of 1990, 6 years after Connector Marsh was constructed, differences still existed between it and the natural marsh. Cordgrass at the constructed marsh had only 60% of the biomass of that at Paradise Creek. Plants were dense but generally shorter than in natural marshes. The vegetation was judged inadequate for support of predators, including beetles that control herbivorous insects and the endangered light-footed clapper rail (Zedler 1993). In 1992, when the marsh was 8 years old, the plant canopy declined; at the same time, damage by scale insects was widespread (Boyer and Zedler 1996). In 1994, 10 years after transplantation, cordgrass canopies were still short (Boyer and Zedler, unpublished data).

5.1.3 Limited Effectiveness of One-Time Soil Amendments

My colleagues and I hypothesized that the rate of development of constructed wetland ecosystems would be accelerated by adding organic matter and nitrogen to the substrate. We tested this hypothesis in a newly constructed salt marsh by augmenting the organic matter and nitrogen in the sediments (Gibson et al. 1994). We expected cordgrass to show increased growth in proportion to the amount of nitrogen added, whether from inorganic or organic sources. We further expected that nitrogen pools would be increased after the addition of amendments (especially amendments with low carbon:nitrogen ratios) and that fertilizer containing inorganic nitrogen would also increase nitrogen pools, but for a shorter period. The field experiment took place within a newly excavated 7-ha marsh, known as Marisma de Naci6n, located along San Diego Bay. Amendments included straw (carbon:nitrogen = 84), alfalfa (carbon:nitrogen = 12), and ammonium sulfate (inorganic nitrogen).

The organic amendments rapidly stimulated growth of cordgrass. Significant effects on plant biomass occurred as early as 2 months after planting. Highest growth was obtained when alfalfa was added, either with or without inorganic nitrogen. The combination of inorganic nitrogen and straw increased growth significantly more than straw alone (Gibson et al. 1994), indicating that the straw-amended system was more nitrogen limited than the alfalfa system (Fig. 5.1). Rototilling appeared to have an adverse effect; biomass was consistently low in the rototilled control areas.

Overall, the growth data indicate that alfalfa caused the most growth because it contained more nitrogen than the other treatments. However, relatively little of the nitrogen added was recovered by the aboveground plant material (Gibson et al. 1994). A substantial fraction of the readily mineralizable nitrogen was leached or denitrified in the month before planting. Results of an assay (decomposition in litterbags) done in 1992 indicated that as much as 80% of the added nitrogen can be mineralized in the first month (Gibson 1992). Because destructive sampling was not compatible with the long-term objectives of the amendment study, we did not estimate nitrogen storage in belowground biomass. Even if roots and rhizomes had eight times the aboveground biomass (Valiela et al. 1982), the fraction of nitrogen retained by the vegetation was small.

In the first year of the study, nitrogen-rich organic matter (alfalfa) increased plant growth but not sediment organic matter and nitrogen pools (Gibson et al. 1994). Thus, it was no surprise that in the second year, plant growth was still limited, even though the effects of treatment were still visible (Fig. 5.1). The highest cordgrass biomass developed in areas amended with alfalfa. However, the maximum biomass in year 2 was only about half that of the standing crop of a natural cordgrass marsh (c.f. data in Winfield 1980).

5.1.4 Why Short-Term Fertilization Does Not Enhance the Nitrogen Pool

In natural marshes, the reserve of mineralizable organic nitrogen accumulates and is maintained by decaying plant material and root exudates. The associated vegetation persists through continuous turnover of nitrogen within the rhizosphere and with new inputs only occasionally (as with flooding). In natural marshes, one-time amendments do not produce this necessary nitrogen pool. No effect of treatment could be shown in the organic carbon pools 5 months (July 1990) or 7 months (September 1990) after amendments were added. The data indicate that most of the organic matter was mineralized, and most of the nitrogen released, in the weeks immediately after addition. These findings were confirmed by the results of a decomposition experiment with buried litterbags (reported in Gibson et al. 1994), and by those of the following, more extensive, experiment, which explored the role of soil texture in retaining inorganic and organic amendments. In the latter experiment, we also tested waste products (kelp wrack, horse manure) as novel ways to enhance the nutrient pool more effectively and more cheaply than by adding alfalfa.

We hypothesized that amendments would be retained more readily by finer than by sandier soils. The field experiment was started in March 1992 in a disturbed area adjacent to Paradise Creek Marsh at an elevation judged suitable for cordgrass. Existing sediments were loamy sand (84% sand, 6% silt, and 10% clay). The texture of the transported sediment was sandy loam to loam (50–62% sand, 25–30% silt, and 10–20% clay). Nitrogen and carbon contents of both the existing and transported sediments were less than 0.1%.

The mass of nitrogen in each of eight amendments was 96 g of nitrogen per square meter. The amendments were as follows: low dose of alfalfa hay, high (double) dose of alfalfa hay, kelp, horse manure, ammonium sulfate, sulfur-coated urea (slow-release nitrogen), alfalfa plus clay, and fast-release nitrogen plus clay. The two controls were clay and sand with no added nitrogen. After the soil amendments were added, soil parameters (total nitrogen, KCl-extractable carbon and nitrogen, porewater carbon and nitrogen, sulfides, redox potential, salinity, and pH) were measured 4 days, 7 days, 2 weeks, 4 weeks,

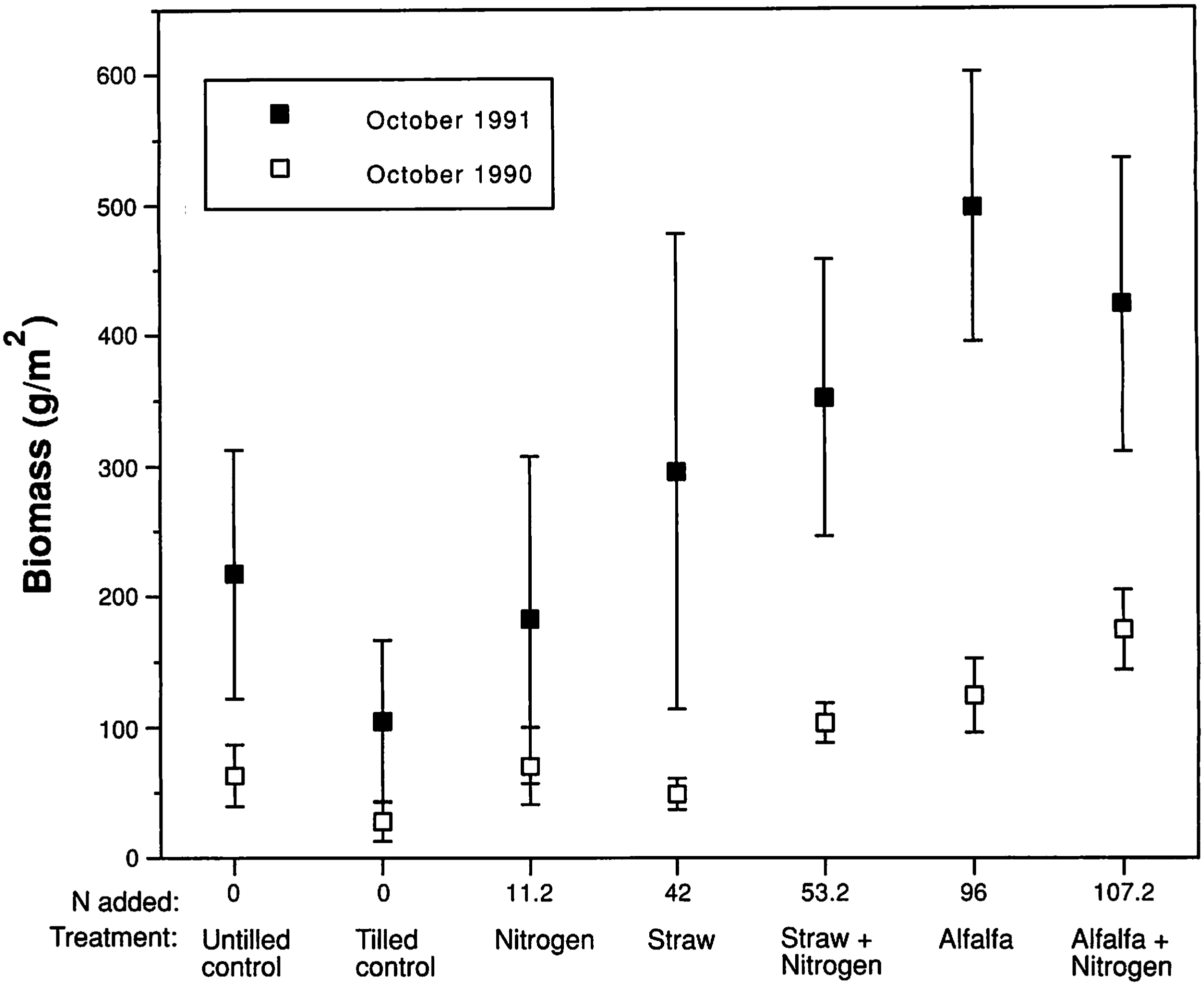


Figure 5.1 Cordgrass biomass (estimated from data on total stem length) during the first and second years after amending soils with organic and inorganic nitrogen. Plants grew better with high addition of nitrogen (N)—*x* axis shows grams of N added—and performed better in year 2 after vegetative expansion; yet the highest standing crops were only half the biomass of marshes in natural cordgrass. Modified from Gibson et al. 1994.

2 months, and 4 months later. Nutrients were measured in both sediments and porewater wells (20 cm deep).

Nitrogen was released in a pulse and was rapidly lost from both fine and coarse soils and for both organic and inorganic amendments, as indicated by total Kjeldahl nitrogen (TKN) levels 4 and 7 days after fertilization (Fig. 5.2). Variability was high and appeared to obscure any effects of treatment; no significant differences were found. At 4 days, soil treated with fast-release nitrogen plus clay had the highest mean TKN (3.6 mg nitrogen per gram of soil), which compared favorably with maximum levels found in local natural marshes: 3.01 mg nitrogen per gram in Paradise Creek (PERL, unpublished data) and 2.38 mg nitrogen per gram in Tijuana Estuary (Langis et al. 1991). However, subsequent TKN levels were less than half the initial levels. Soil TKN showed no patterns after the first week. Soil texture did not appear to have an effect on nitrogen retention, suggesting that higher concentrations of clay are needed. In all cases, the total amount of nitrogen present 1 year after treatment was considered inadequate from a biological perspective.

5.1.5 Producing Tall Cordgrass Vegetation with Multiple Additions of Nitrogen

Katharyn Boyer and Joy Zedler

Because the one-time soil amendment (Gibson et al. 1994) was insufficient to produce canopies in a constructed marsh equivalent to those in a reference marsh, an experiment was done in the constructed Connector Marsh to examine the effect of extending the period of fertilization. We used biweekly applications of urea for as little as 1 month and as long as 6 months. Results were examined at the end of the first growing season (1993).

We continued one fertilizer treatment for a second year (1994). One set of plots fertilized for 6 months in 1993 received the same number of applications during 1994, whereas another set was not fertilized in the second year. We hypothesized that the effects of treatment during the first year would be retained through accumulation of nutrients in the soils or storage in belowground tissue or, alternatively, that fertilization for 1 year might not be sufficient to sustain tall cordgrass vegetation. At

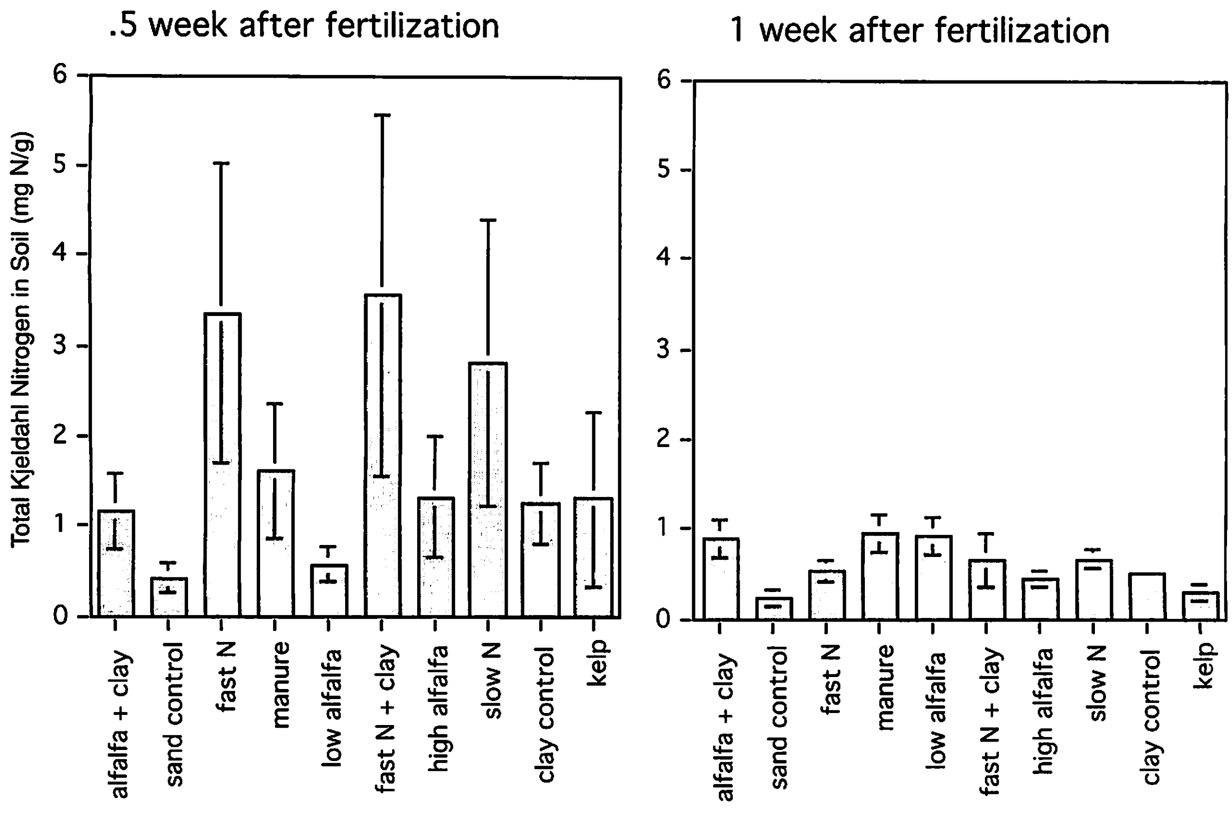


Figure 5.2 Mean total Kjeldahl nitrogen (TKN) in soil after inorganic and organic nitrogen additions. Number of samples = 4; error bars represent ± 1 S.E.

the same time, we compared short-term fertilization in 4 different months. Statistical analyses were based on a total of 10 treatments and seven replicates (Boyer and Zedler, unpublished data).

The vegetation responded significantly to the addition of nitrogen. In 1993, additions of nitrogen caused increases in both aboveground biomass and foliar nitrogen in Connector Marsh. Late in the season, cordgrass in the plots treated with 12 applications of fertilizer had the highest total stem length. Next, in order, were the plots treated with 8 and 4 applications. The plants in the control plots had the lowest total stem length. In all cases, mean foliar nitrogen correlated with the amount of nitrogen added. Plants in plots treated only twice, with urea applied at different times during the growing season, did not differ in total stem length. For aboveground growth, the duration of fertilizer treatments (total amount added) was more important than the timing of the additions.

After being fertilized, the cordgrass canopy was tall and robust by August and September in 1993. All fertilized plots met the criteria (Zedler 1993) for cordgrass canopies suitable for nesting by clapper rails, whereas the control (unfertilized) plots did not. However, just 1 year later (August 1994), only the set of plots fertilized in both 1993 and 1994 had enough tall stems to meet the canopy height criteria. Thus, belowground biomass did not store sufficient nitrogen to produce a tall canopy the following year, even when nitrogen was applied biweekly throughout the growing season. Nor was soil nitrogen (total Kjeldahl N and KCl-extractable N) increased with fertilization in either year, supporting our hypothesis that nitrogen is poorly retained in the coarse soils of the constructed marsh.

Cordgrass growth was generally less robust in 1994 than in 1993. Greater growth in 1993 in both fertilized and control plots was probably associated with the January 1993 flood, which both lowered soil salinity and added nutrients. Stem lengths and densities were also greater in 1993 than in 1994 in the natural marsh at Paradise Creek. The lack of a response to fertilization at Paradise Creek in 1993 was further evidence of the substantial effects of flooding. Addition of nitrogen had no effect on stem density, stem heights, or foliar nitrogen in 1993, suggesting that the cordgrass of this natural marsh was not nitrogen limited immediately after the flood. Because this conflicts with results of other studies of fertilization of plants in a natural marsh (e.g., Covin and Zedler 1988), we think that the flood provided a pulse of nitrogen and low salinities in early 1993, ameliorating the usual limitation by nitrogen. Similar flood effects have been reported for Tijuana Estuary (Zedler et al. 1992).

In summary, nitrogen fertilization for a 6-month period during a flood year produced cordgrass canopies that were comparable to those in natural marshes, but this effect was not sustained. At the end of the second growing season, cordgrass canopies were not suitable for nesting by clapper rails except in areas that were fertilized in both years.

5.1.6 Impacts of Nitrogen Fertilization on Insect Pests

Katharyn Boyer and Joy Zedler

In conjunction with the experiment just described, we assessed the response to fertilization of an existing population of scale insects (*Haliaspis spartina*, Homoptera: Diaspididae) feeding on the leaves of cordgrass in the constructed marsh (Boyer 1994; Boyer and Zedler 1996). Because many studies have shown that fertilization increases the damage caused by herbivores (Pfeiffer and Wiegert 1981, McNeill and Southwood 1982, Bryant et al. 1987, Lightfoot and Whitford 1987, Strauss 1987, Johnson 1991), we were concerned that fertilizing at Connector Marsh might exacerbate this infestation. Alternatively, others have found that environmentally stressed plants are more susceptible to major damage by insects (Knerer and Atwood 1973, Mattson and Addy 1975, White 1976, Redak and Cates 1984, White 1984, Mattson and Haack 1987, Louda 1988, Larsson 1989). Low amounts of nitrogen in the soils in the constructed marshes of Sweetwater National Wildlife Refuge are likely to be an environmental stress for cordgrass, possibly allowing an increase in the population of *Haliaspis*. Stressed plants may be more nutritious because of greater availability of soluble nitrogen and carbohydrates (White 1969, 1974, 1976; Rhoades 1979, 1983). Also, stressed plants, with limited energy and nutrient resources, may allocate less energy to defense (Rhoades 1983), both chemical and mechanical, allowing growth of populations of phytophagous insects.

Once juvenile insects were observed on the leaves of new cordgrass shoots (mid-April, 1993), we began making biweekly estimates of *Haliaspis* densities. We used abundance classes to estimate the number of *Haliaspis* per stem: class 0 = 0, 1 = 1–9, 2 = 10–99, 3 = 100–999, and 4 = 1000 or more. Although dispersal occurred in two main pulses in 1993 (late May and late July), our hypothesis that timing of *Haliaspis* dispersal might influence the result of fertilization was not supported.

Contrary to our prediction, nitrogen fertilization did not exacerbate *Haliaspis* damage to cordgrass in the constructed Connector Marsh. Long-term fertilization (10 applications over 20 weeks) appeared to reduce *Haliaspis* densities relative to the density in control plots by August. Although early-season establishment of *Haliaspis* was greater in the plots treated with 12 applications of fertilizer (6 applications at that time) than in the controls, this trend in abundance reversed by August, when the control plots contained many more stems infested with large numbers of *Haliaspis* than the long-term fertilized plots did (Fig. 5.3) Mean abundance per stem also became lowest in the long-term fertilized plots.

Lower abundance of *Haliaspis* by the late season in the fertilized plots may have been due to one or more of the following: increased attack by predatory insects that use tall plants as a high-tide refuge; dislodging of *Haliaspis* (particularly juveniles) through abrasion of sharp-edged leaves; more difficult stylet penetration and prolonged attachment on hardened leaf tissue; or relief from the stress caused by nitrogen limitation, possibly leading to lower availability of soluble nitrogen and carbohydrates or increases in chemical defenses.

Plots fertilized only in March, April, June, or August did not differ from controls in mean abundance of *Haliaspis*. The insects were never abundant in the fertilized or control plots in the adjacent natural marsh. Concerns that cordgrass fertilization might provoke damage by scale insects were allayed, but other herbivores complicated our study in 1994. Patchy damage by lepidopteran stem-boring larvae and their predators (small mammals) occurred in the experimental plots in 1994, beginning in July. Although control plots showed the least variability and plots treated both seasons the most, no plots were exempt from damage (Boyer and Zedler, unpublished data). Other areas remote to our fertilization study were also damaged, in both constructed and natural marshes; therefore, we lack direct evidence that this herbivory was promoted by fertilization of cordgrass.

This study suggests that fertilization of constructed salt marshes in San Diego Bay may proceed without concern that *Haliaspis* outbreaks will be facilitated. Other problems may develop, especially if larger areas are fertilized and different consumers are attracted to the plots. Our experimental program has been expanded to test the effect of fertilizing patches of cordgrass that are large enough to attract clapper rails to nest. To determine how long fertilization will be required, we began an experiment in 1995 that involves fertilizing plots of cordgrass for 1–5 years. We will compare plants, soils, and herbivory at the end of each growing season.

5.1.7 Recommendations for Improving Restoration of Cordgrass

Katharyn Boyer and Joy Zedler

Because we found that nitrogen was not retained in the soils after the first season of fertilization and that belowground storage was inadequate to sustain tall canopies of cordgrass through the second year, long-term corrective action is needed. We recommend using continual fertilization over multiple growing seasons to sustain tall canopies, as required in the mitigation agreement (U.S. Fish and Wildlife Service 1988).

For future projects, we recommend that restoration sites be provided with a surface layer of fine sediments. Because it is so difficult to create suitable habitat for nesting by clapper rails in areas with sandy soils, we recommend salvaging and reusing fine-textured soils from the areas that will be lost to development. When restoration projects are undertaken as part of a mitigation program, the marshes that are to be damaged should

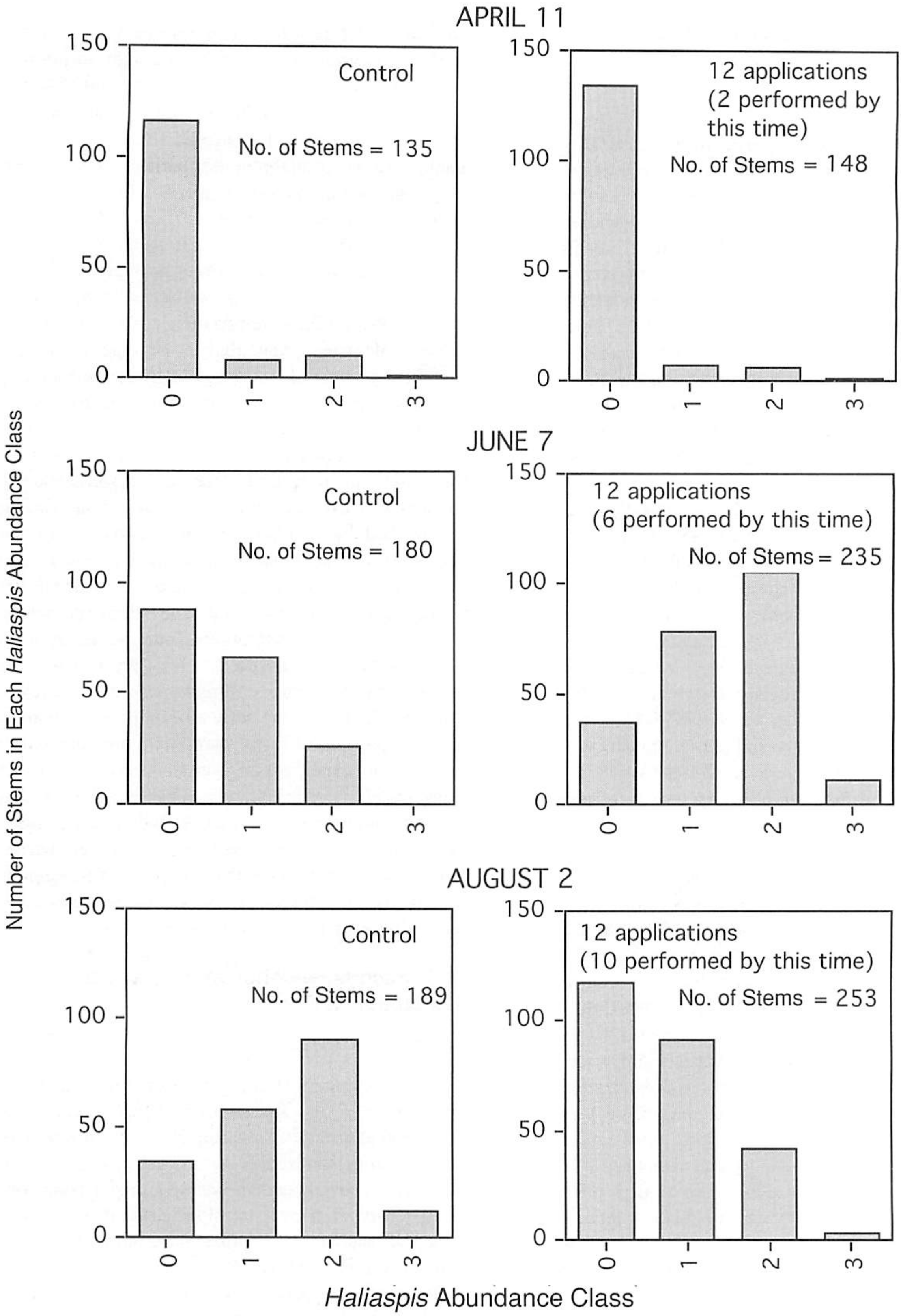


Figure 5.3 Number of cordgrass stems that supported different abundances of *Haliaspis* (class 0 = 0 insects, 1 = 1–9, 2 = 10–99, 3 = 100–999, 4 = 1000 or more). The control and the 12-application treatments were compared in April, June, and August 1993. The total number of stems sampled in seven 0.10-m^2 quadrats is shown in the upper right of each graph

have a comprehensive salvage plan, so that plants, animals, and soils can be moved to the mitigation site.

The responses of herbivorous insects to nitrogen amendments need to be evaluated when large areas are fertilized. It is unclear whether the patchy damage caused by stem-borers or mammals in our experiment was related to fertilization, and such damage needs to be closely watched. Further corrective action may be needed to curtail outbreaks of pest species.

Our experience with the 10-year-old constructed marsh shows that it is difficult to compensate for damages to clapper rail habitat, especially on coarse substrates. Other factors may limit use of this mitigation site by clapper rails, such as the adjacent freeway, contaminants in street runoff, dogs and cats that can prey on birds, and power lines which attract raptors that can prey on rail chicks. Because there is no guarantee that habitat useful to clapper rails can be created in urban settings, we recommend no further damage to the few remaining marshes that are used by this endangered species.

5.2 EXPERIMENTAL APPROACHES WITH MESOCOSMS

John Callaway and Joy Zedler

A wide variety of techniques might be used to accelerate the development of ecosystems in restoration sites, but our work with nitrogen amendments suggests that preliminary experimentation is needed to develop successful methods.

5.2.1 The Value of Experimentation Before Restoration

Rigorous evaluation of restoration techniques is often difficult if done on a large scale in actual restoration sites. The cost of trying different methods will be high if the area is large; also, harvesting plants or trapping animals to assess experimental treatments may conflict with the goal of creating habitat. Thus, most attempts at restoration are large-scale, unreplicated trials, and statistical analysis of the results is not possible. Because of the constraints in evaluating techniques on a large scale in restoration sites, it is best to work out new techniques in a smaller scale experimental setting before restoration. Field conditions can be approximated more closely by using mesocosms (medium-size, outdoor experimental units) than by using laboratory experiments. Testing new techniques in mesocosms may indicate pitfalls before the techniques are applied to large areas with great cost and little benefit.

5.2.2 The Tidal Mesocosms at Tijuana Estuary

A new facility at Tijuana Estuary has allowed us to evaluate the usefulness of mesocosms for testing how hydrology affects the development of mid-intertidal salt marsh. We planned the mesocosms to support pickleweed because it is the most widespread dominant plant in salt marshes in Southern California. We varied two factors, tidal flushing and freshwater inflow, which determine the degree of water circulation, the salinity of channel waters and marsh soils, and, ultimately, the composition of the plant and animal communities. Our hydrologic treatments mimicked conditions typical of mid-intertidal marshes throughout Southern California, including natural (fully tidal) and modified coastal wetlands that were either (a) impounded (i.e., subject to tidal flooding and freshwater inflows but not able to drain) or (b) excluded from tidal action (i.e., with freshwater inputs and ability to drain, but with tidal inflows precluded).

The 24 mesocosms were planted with pickleweed and irrigated with fresh water to aid establishment. When pickleweed achieved 80% cover in all mesocosms, freshwater inflows were terminated and the tidal treatments were initiated. Eight mesocosms were left open to full tidal flushing, 8 mesocosms were fitted with tide gates that were opened on incoming tides and closed at high tide to impound seawater, and the remaining 8 mesocosms were fitted with a gate that excluded tidal inflows and a one-way valve that discharged any runoff. During the last 6 months of the experiment, fresh water was added to 4 mesocosms in each group of 8, resulting in 6 experimental treatments: 3 tidal regimes, each with and without fresh water.

5.2.3 Determining Factors That Limit Ecosystem Development

An unexpected problem that we encountered with the mesocosms provides a lesson for both future research and restoration projects. The substrate used to construct the mesocosms was unsuitable for rapid development of a pickleweed canopy. The mesocosms were located in an area of old fill material, and the soil had a much coarser texture (about 60% sand, 20% silt, and 20% clay) than that in the natural marsh (less than 10% sand, about 20% silt, about 70% clay). Because of the coarse substrate, subsurface drainage in the mesocosms was much greater than in natural salt marshes. The mesocosms that were designed to impound water drained easily via subsurface seepage (Fig. 5.4), and water levels were similar to those in areas with tidal action, where water was removed via surface drainage.

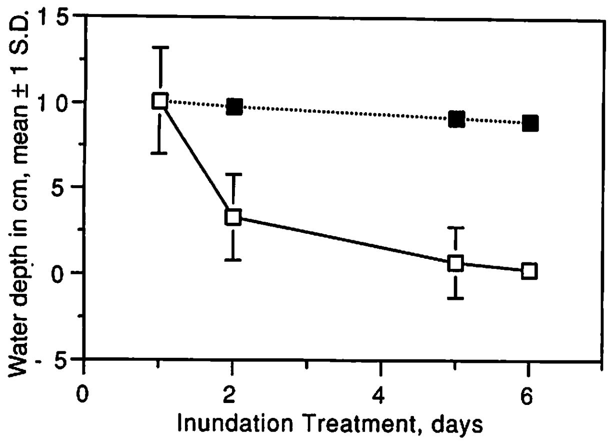


Figure 5.4 Comparison of actual mesocosm drainage with expected impoundment curve. —■— expected —□— observed. Data collected by J. Boland, PERL.

Natural salt marshes are characterized by fine-grained sediments, which inhibit drainage and create anaerobic conditions. Anaerobic conditions slow the rate of decomposition and increase accumulation of soil organic matter which is important to many wetland functions. The accumulation of organic matter further slows drainage because organic soils have high water-holding capacity. The magnitude of the effect of improper substrate became clear in the mesocosms, and the importance of subsurface drainage was documented.

The problem of excessive drainage indicates the importance of considering substrate conditions when planning wetland restoration projects. Without the proper substrate, it will be difficult to restore important wetland functions. Substrate considerations have been evaluated in other wetland restorations (Craft et al. 1988, Langis et al. 1991); however, the focus has been on the low marsh, in areas dominated by cordgrass, and on nutrient concentrations and the amount of organic matter present. Subsurface drainage is more critical at higher elevations, where tidal inundation is less frequent and soils are subject to drying and hypersalinity.

A second problem that exacerbated the effects of poor water retention was due to the intertidal position and location of the mesocosms. The mesocosms were excavated to 6.1 ft (1.9 m) MLLW, and they should have been inundated by many high tides during the summer and winter extremes. However, the mesocosms are far from the mouth of the estuary, which means that tidal ranges are attenuated, and fewer tides than expected inundate the marsh. We had to pump tidal water into the mesocosms to mimic the desired inundation regime (wetting during all predicted tides over 6.1 ft MLLW). Obviously, locating a restoration site far from the tidal source means that elevations will have to be lower than those of sites near the ocean inlet to benefit from the same hydroperiod (i.e., same frequency and duration of tidal inundation).

5.2.4 Consequences of Coarse Substrate

The first consequence of coarse substrate was difficulty in establishing pickleweed. Where substrates are suitable, this species is a good invader, as the "weed" part of its name suggests. We did not anticipate difficulty in getting pickleweed to grow from either seed or transplants. However, it took repeated transplantation through early summer to develop the desired canopy (about 80% cover) by fall. High salinities (50–60 ppt) and low soil moisture were likely causes.

Soil salinities in both the impounded and tidal mesocosms became quite low in winter and extremely high in summer (Fig. 5.5). Where tides were excluded, hypersalinity was even more extreme, because there was little opportunity for surface discharge and salt removal. These results suggest a generalization that coarse soils increase the range of salinities that a wetland will experience. Low salinities can encourage invasions of exotic species, and hypersaline conditions stress even the most salt-tolerant halophytes.

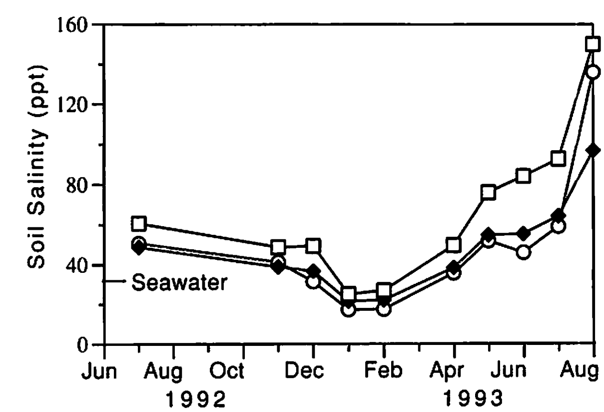


Figure 5.5 Soil salinity of mesocosms under tidal treatments. N = 8 mesocosms per treatment through June 1993, n = 4 thereafter. —□— tides excluded —◆— fully tidal —O— tides impounded.

Over time a restored wetland is likely to accumulate fine-grained material because of the slow water velocities that are typical of wetland hydrology, and this may lessen the initial differences in texture in surface substrates. However, the rate of accumulation of sediments in tidal wetlands is about 0.1–1.0 cm/yr, so changes in texture will be slow. Organic matter and nutrients will also accumulate in the sediments; however, the rate of accumulation of both is likely to be slower in coarsely grained sediments. Wetland functions will be lost during this period of accumulation.

Because coarse soils increase subsurface drainage, increase the range of soil salinities (leading to extreme hypersalinity in summer), and slow the establishment of pickleweed, it follows that providing the appropriate soil texture will accelerate marsh development.

5.2.5 Conclusion and Recommendations

Our findings call attention to the importance of providing substrates with appropriate soil texture at restoration sites, especially in the high marsh. In the future, restoration plans need to consider the condition of the proposed substrate, whether it is subsurface material to be excavated, dredge spoils, or other materials. We recommend that any natural marsh sediments that are destined to be damaged through development be removed and stockpiled for later reuse in the mitigation site. This is likely to be the most suitable method of ensuring proper substrate conditions if the mitigation site has coarse soils. Otherwise, significant amendment of existing substrates may be necessary, and this is not always successful (see section 5.1).

5.3 ARTIFICIAL WETLANDS TO AUGMENT USE BY ESTUARINE BIRDS

Joy Zedler and Barbara Kus

The value of natural wetlands to bird populations is well recognized, and declines in waterfowl numbers are often attributed to losses in wetland area. If the destruction of wetland reduces bird populations, then adding wetland habitats might improve the situation. This idea was tested at Tijuana Estuary in the late 1980s.

5.3.1 Construction of Shallow Wetlands from Upland at Tijuana Estuary

Freshwater wetlands are thought to have occurred historically near saline wetlands all along the coast of Southern California (Swift et al. 1993). Few of these wetlands remain, and it is unclear what role they played in supporting resident and migratory birds. We hypothesized that providing shallow freshwater wetlands near Tijuana Estuary would lead to an increase in the populations of birds using the estuary, by providing a different type of habitat, more open water, and refuges during high tides.

Funding from the State Resources Agency, Environmental License Plate Fund, and the State Coastal Conservancy made it possible to test the habitat-augmentation concept in a general manner. However, we could not determine whether the birds that used the artificial wetlands came from the nearby estuary (i.e., a local shift in habitat use) or from elsewhere (i.e., new birds were attracted to the area). Because shallow, standing fresh water is rare at Tijuana Estuary, we suspect that bird use was proportional to the amount of habitat made available and that the wetlands attracted more birds to the local region than would otherwise have occurred here. Only a long-term study of both the estuary and the artificial wetlands, including banded birds, could determine how much local bird productivity could be enhanced by wetland construction.

Between 1986 and 1987, more than 70 wetlands were constructed to the southeast of Tijuana Estuary within a 28-ha site at 1800 Monument Road.The site had previously been used for agriculture, but aerial photos showed it had been vacant since 1967. The land had been purchased for the National Estuarine Research Reserve and deeded to the City of San Diego, which provided a long-term entry permit to PERL.

Shallow basins (depth, 0.3–0.6 m) were excavated in sandy sediment that had accumulated from previous floods. All basins were lined with construction-grade plastic, partially refilled with soil, and watered with an irrigation system. In addition, an estuarine channel west of 1800 Monument Road was deepened to allow pumping of brackish water into selected wetlands. The artificial wetlands differed in size, salinity, and amount of emergent vegetation. Comparison of bird uses in the wetlands allowed us to determine what wetland construction techniques work best to attract different bird species.

Construction of the artificial habitat occurred in four phases. First, we used a Bobcat to excavate three small wetlands in the spring of 1986 to work out techniques. Second, after locating a source of larger and stronger plastic, we excavated 15 small (6.1×30.5 m) wetlands in the summer and fall of 1986. In the third phase, we constructed 54 small wetlands (6.1×30.5 m) and 18 large wetlands (30.5×30.5 m); these required a bulldozer and were completed in April 1987. Maximum size was determined by the size of plastic liners that could be obtained and manipulated. Because both the smaller and larger wetlands were susceptible to canopy closure by emergent plants, we pumped estuarine water into one-third of the freshwater wetlands, allowing it to evaporate and create hypersaline conditions that reduced emergent vegetation. Using this technique, in phase four we created one large (0.8 ha) unvegetated wetland with a broad, unvegetated shoreline.The project was conducted by Chris Nordby, Barry Dubinski, and other PERL staff members; the wetlands were constructed, studied, and maintained for 3 years at a total cost of about $123,000.

5.3.2 Attributes of the Artificial Wetlands

Wetland vegetation developed in all the impoundments soon after water was added (PERL, unpublished data). *Typha domingensis, Scirpus robustus,* and *S. olneyi* all invaded on their own; *Typha* was the most aggressive colonizer. Submergent vegetation included *Potamogeton filiformis, P. latifolius,* and two genera of macroalgae: *Chara* sp. (the stonewort) and *Nostoc* sp. (a bluegreen alga that forms gelatinous colonies or irregular balls). *Nostoc* species fix nitrogen.Vegetation became dense in much of each wetland, even though the only nutrient sources were the soil and rainfall. Later, birds probably contributed additional nutrients.

Animals that colonized the water were counted to determine whether food was present for use by birds. Using dip nets and light traps, we detected a diversity of invertebrates in the water column and in the benthic substrate (more than 100 genera, PERL, unpublished data). Zooplankton included cladocerans and copepods. Insects included mayflies, damselflies, dragonflies, water boatmen, back swimmers, water striders, whirligig beetles, predatory diving beetles, mosquitoes, and midges. Physid snails were abundant in several wetlands.

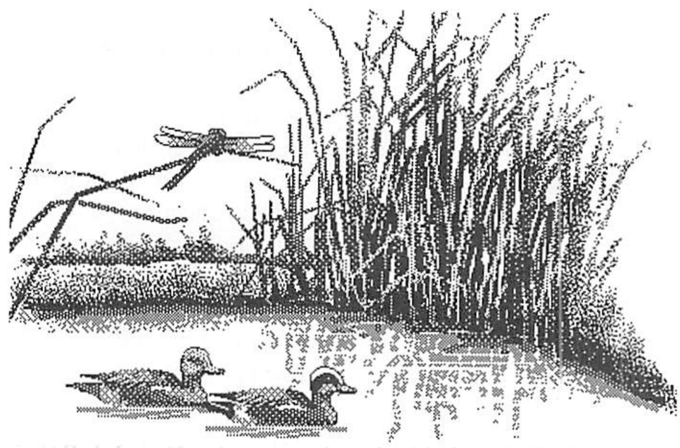
Four insect species commonly found in the artificial wetlands are indicators of specialized, clean-water habitats. The predatory diving beetles *Rhantus* sp. and *Agabus disintegratus* and the back swimmer *Buenoa* sp. are rare in the region because of the poor water quality (sewage-laden river water, street runoff, and agricultural runoff). Yet they became abundant in the larger freshwater wetlands. *Trichocorixa reticulata,* the salt marsh water boatman, was found in high densities at the saline mud flat. It is an indicator of healthy salt marsh and wetland habitat (C. Nagano, U.S. Fish and Wildlife Service, personal communication). The occurrence of these important predatory species is notable, and the lack of mosquito outbreaks may be due to their abundance.

In December 1987, we added mosquito fish (*Gambusia affinis*) to 10 of the large wetlands to provide food for the larger wading birds, and in June 1988, we added large-mouth bass (*Micropterus salmoides*) to the same wetlands.

5.3.3 Use of the Artificial Wetlands by Water Birds

Birds visiting the wetlands were surveyed approximately weekly from June 22, 1987, through May 30, 1988, by using established routes and two 2-m-tall platforms. Overall, 32 species of water birds and 30 species of terrestrial birds were observed using the wetlands (Table 5.1). Some species appeared as soon as the wetlands were constructed, and they remained throughout the year of study. Abundant in this group were mallards (*Anas platyrhynchos*), coots (*Fulica americana*), herons, egrets, and snipe (*Gallinago gallinago*). Waterfowl showed a second pattern of use: they arrived in the late summer and early fall, stayed over the winter, and departed in the spring. These included pintails (*Anas acuta*), American widgeon (*Anas americana*), green-winged teal (*Anas crecca*), northern shovelers (*Anas clypeata*), ruddy ducks (*Oxyura jamaicensis*), and buffleheads (*Bucephala albeola*). Shorebirds such as sandpipers (*Calidris* sp.) probably also belonged to this group, but they were too infrequent to discern a pattern. As a third group, local resident shorebird species (black-necked stilts, killdeer, and American avocets) became abundant in spring and nested and produced chicks around the artificial wetlands.

The larger wetlands were highly attractive to water birds; ducks were the most frequent and abundant users (PERL, unpublished data). The wetlands that were



consistently most attractive to birds had substantial blooms of macroalgae (more than 60% cover), perhaps as a result of bird use and rapid nutrient turnover. The birds were indifferent to the presence of fish; piscivorous species that were expected to be attracted to ponds with fish were not often present and were never abundant. The smaller wetlands became overgrown with cattails and bulrushes and were less used by water birds. Of the 32 species sighted, only 19 (59%) were seen in the small wetlands. Those found in the smaller wetlands were primarily solitary waders (12 species, with occasionally a coot or a green-winged teal). Although little used directly, these wetlands may have provided insects that were consumed by birds elsewhere.

The 0.8-ha saline mud flat attracted additional and significant bird species to the PERL wetland complex. Whereas ducks were the primary users of the large freshwater ponds, shorebirds commonly used the mud flat (PERL, unpublished data). Avocets, stilts, and killdeer nested at the mud flat and elsewhere where low cover was available. A berm between the two halves of the mud flat resembled an island and was a preferred resting site. These shorebirds had not been seen in the smaller ponds, and we conclude that the broad open areas were the main attraction. Neither exotic plants nor dense vegetation from native species became a problem at the saline mud flat. Invertebrates, initially brought in with the estuarine water, provided a source of food for the estuarine birds.

5.3.4 Conclusions

On the basis of our experiments and observations, we drew the following conclusions:

- Artificial wetlands are used by populations of native birds; they provide additional habitat and refuge during times when native habitats are unavailable (extreme high tides, sea storms, floods, etc.). The wetlands also provide low-cost artificial habitats that can support native plant, invertebrate, and bird species.
- Larger wetlands attract more water birds than smaller ones do. Ducks are the most numerous users of shallow, open-water habitats.

Table 5.1. Bird Use of Artificial Wetlands at Tijuana River National Estuarine Research Reserve

Common Name	Latin Name	No. of Sightings	% of Surveys Present
WATER BIRDS			
Grebes (2 sp.)	**Total**	**5**	**9.3**
Pied-billed grebe	*Podilymbus podiceps*	1	2.3
Horned grebe	*Podiceps auritus*	4	7.0
Ducks (12 sp.)			
Dabbling ducks (8 sp.)	**Total**	**1798**	**95.4**
Mallard	*Anas platyrhynchos*	596	86.0
Northern pintail	*Anas acuta*	260	48.8
Gadwall	*Anas strepera*	18	11.6
American widgeon	*Anas americana*	120	25.6
Blue-winged teal	*Anas discors*	2	2.3
Cinnamon teal	*Anas cyanoptera*	160	53.5
Green-winged teal	*Anas crecca*	578	30.2
Northern shoveler	*Anas clypeata*	64	16.3
Diving ducks (4 sp.)	**Total**	**162**	**51.2**
Ring-necked duck	*Aythya collaris*	3	4.7
Lesser scaup	*Aythya affinis*	4	7.0
Ruddy duck	*Oxyura jamaicensis*	42	25.6
Bufflehead	*Bucephala albeola*	113	41.9
Rails and Coots (2 sp.)	**Total**	**2171**	**74.4**
Virginia rail	*Rallus limicola*	2	4.7
American coot	*Fulica americana*	2169	69.8
Herons and Egrets (6 sp.)	**Total**	**23**	**37.2**
Great blue heron	*Ardea herodias*	12	20.9
Great egret	*Ardea alba*	1	2.3
Cattle egret	*Egretta ibis*	2	2.3
Snowy egret	*Egretta thula*	5	7.0
Green-backed heron	*Ardeola striata*	2	4.7
American bittern	*Botaurus lentiginosus*	1	2.3
SHOREBIRDS			
Stilts and Avocets (2 sp.)	**Total**	**266**	**62.8**
American avocet	*Recurvirostra americana*	34	27.9
Black-necked stilt	*Himantopus mexicanus*	232	53.5
Plovers (1 sp.)	**Total**	**119**	**62.8**
Killdeer	*Charadrius vociferus*	119	62.8
Sandpipers and Snipes (7 sp.)	**Total**	**130**	**51.2**
Western sandpiper	*Calidris mauri*	1	2.3
Least sandpiper	*Calidris minutilla*	16	9.3
Yellowlegs	*Tringa* sp.	6	11.6
Spotted sandpiper	*Tringa macularia*	2	4.7
Marbled godwit	*Limosa fedoa*	2	4.7
Dowitcher	*Limnodromus* sp.	91	27.9
Common snipe	*Gallinago gallinago*	12	16.3
LAND BIRDS*			
Raptors (7 sp.)			
Osprey	*Pandion haliaetus*		2.3
Black-shouldered kite	*Elanus leucurus*		44.2
Northern harrier	*Circus cyaneus*		79.1
Cooper's hawk	*Accipiter cooperii*		4.7
Red-tailed hawk	*Buteo jamaicensis*		32.6
Golden eagle	*Aquila chrysatos*		4.7
American kestrel	*Falco sparverius*		32.6

Common Name	Latin Name	No. of Sightings	% of Surveys Present
Perching Birds (20 sp.)			
Songbirds (17 sp.)			
Barn swallow	*Hirundo rustica*		18.6
Cliff swallow	*Hirundo pyrrhonota*		32.6
Loggerhead shrike	*Lanius ludovicianus*		9.3
Bushtit	*Psaltriparus minimus*		4.7
House wren	*Troglodytes aedon*		2.3
Bewick's wren	*Thryomanes bewickii*		2.3
Water pipit	*Anthus spinoletta*		4.7
Yellow-rumped warbler	*Dendroica coronata*		2.3
Lesser goldfinch	*Carduelis psaltria*		2.3
House finch	*Corpodacus mexicanus*		11.6
Common yellowthroat	*Geolypis trichas*		46.5
Yellow-breasted chat	*Icteria virens*		9.3
Western meadowlark	*Sturnella neglecta*		25.6
Red-winged blackbird	*Agelaius tricolor*		20.9
Brown-headed cowbird	*Molothrus ater*		2.3
Song sparrow	*Zonotrichia melodia*		55.8
White-crowned sparrow	*Zonotrichia leucophrys*		4.7
Flycatchers (3 sp.)			
Black phoebe	*Sayornis nigricans*		58.1
Say's phoebe	*Sayornis saya*		11.6
Western kingbird	*Tyrannus verticalis*		4.7
Other (3 sp.)			
Mourning dove	*Zenaida macroura*		20.9
Anna's hummingbird	*Archilochus anna*		2.3
Belted kingfisher	*Ceryle alcyon*		4.7

Note: Birds were observed weekly from June 1987 through May 1988. Data are total individual birds sighted and percentage of surveys when the birds were sighted. Group percentages are for surveys when one or more birds from that category were sighted.

*Only presence–absence data were recorded for landbirds.

- Cattails invade readily and dominate shallow freshwater wetlands. Emergent vegetation rapidly closes the canopy in shallow ponds. Exotic plants can become a problem in artificial freshwater wetlands.
- Saline mud flats are easily constructed if an initial source of saline water is available. The salt becomes concentrated via evaporation and prevents establishment of vegetation. The artificial hypersaline mud flat successfully attracted shorebirds and other water birds.
- Artificial wetlands require continued maintenance. They are not self-sustaining. Needed are a water supply and occasional checks for damage to the plastic liners. Such maintenance is not costly, but it is necessary to maintain the habitat.

Artificial wetlands could be used to increase the types of habitat available inland of natural estuarine wetlands. Construction of saline mud flats on disturbed upland habitat near intertidal wetlands would offer a nontidal feeding and resting ground for shorebirds. Such habitat could be very valuable during high tides.

CHAPTER 6

Improved Methods for Assessing Ecosystem Development

6.1 MONITORING AND INDICATORS OF CONDITIONS THAT SUPPORT BIODIVERSITY

One overwhelming conclusion emerging from past restoration efforts is that it takes many years for a site to evolve into a fully functioning, self-sustaining ecosystem. No system can claim to have done so. Virtually every site that has been monitored continues to change 5–15 years after the initial restoration work. Although change is a characteristic of natural ecosystems, a constructed wetland can be considered functionally equivalent to a natural wetland only if the constructed wetland is changing primarily in response to environmental variations, rather than to shortcomings in the construction—for example, transplanted cordgrass can expand its distribution as suitable space is filled via vegetative expansion, but natural stands can also expand if prolonged flooding makes new areas hydrologically suitable.

Monitoring to distinguish developmental and natural changes should be a matter of detecting directional changes. Expansion of cordgrass after prolonged flooding may be followed by a period of shrinkage as the hydrologic cause reverts to pre-flood conditions. Yet some natural events lead to directional change. Examples are accelerated rise in sea level and subsidence (which would allow inland expansion of cordgrass) and major sedimentation events (which would cause shrinkage).

Monitoring constructed wetlands to determine whether they have achieved functional equivalency with natural systems or meet performance standards presents several challenges, because little background information is available on the basic functions of ecosystems. The food webs have not been well characterized or quantified. The distributions of most species are only generally mapped, and the status of their populations is unquantified. The responses of even the dominant species of plants and animals to environmental norms or extremes are, for the most part, unclear. The effects of many invading exotic species, such as the newly arrived yellowfin goby (a relatively large predator) and a Japanese mussel (now dominant in many subtidal areas of San Diego and Mission bays), are mostly unknown.

Various monitoring programs have been undertaken by PERL (Table 6.1) to track selected environmental conditions and biota at San Elijo Lagoon, Los Peñasquitos Lagoon, Sweetwater Marsh National Wildlife Refuge (which includes natural and constructed wetlands), and Tijuana Estuary. Funding ($18,000–$50,000/year) is inadequate to sample all ecosystem attributes often enough to characterize changes in functioning. Although the most important controlling factor for wetland ecosystems is hydrology, no measurements are made of flow volumes or velocities from either the watershed or ocean sources, and there are no U.S. Geological Survey stream gauges immediately upstream of the estuary (one just upstream of Tijuana Estuary washed out in the 1980 flood and has not been replaced). Nor are water circulation patterns evaluated. The basic biological processes of the system (i.e., primary productivity of vascular plants and algae, nitrogen fixation, decomposition, herbivory, and carnivory) are not monitored, and the growth rates of animal populations are not measured. Instead, surrogates of functioning (structural attributes of each individual site) are evaluated.

6.1.1 What Is Monitored and Why

At each site, the sampling program has been tailored to accomplish the most appropriate work for the money available (Table 6.2). In some cases, additional monitoring work is carried out by agency personnel (e.g., at San Elijo Lagoon, county staff sampled the birds and vegetation). Certain attributes are not management targets in all systems and hence are not sampled (e.g., cordgrass is of concern at Sweetwater Marsh and Tijuana Estuary; Coulter goldfields are tracked at Los Peñasquitos and San Elijo Lagoon).

Frequency of measurement differs for many of the factors assessed, but in general water quality is measured biweekly or monthly, vegetation in September, salinity of marsh soil in April and September, and fishes and invertebrates on a quarterly basis. Special interest species, such as resident birds and rare annual plants, are censused during their reproductive periods (Belding's Savannah sparrows are censused during their territorial nesting

Table 6.1. Southern California Coastal Wetlands Monitored by Pacific Estuarine Research Laboratory

Wetland and Tidal Condition	Funding Agency	Principal Management Questions
San Elijo Lagoon Usually nontidal	County of San Diego	How should the lagoon be managed? When should the mouth be bulldozed open?
Los Peñasquitos Lagoon Often nontidal	Los Peñasquitos Lagoon Foundation (nonprofit)	How can biodiversity be sustained? How can resources be enhanced? How can fish kills be avoided? When should the mouth be bulldozed open?
Tijuana Estuary Usually tidal	National Oceanic and Atmospheric Administration, Sanctuaries and Reserves Division	What are the dynamics of Tijuana Estuary? What are the effects of continuous sewage flows from Tijuana and intermittent flows from sewage spills? Will the estuary recover once sewage flows cease? How should the 200-hectare restoration plan be implemented? How well will the system respond to restoration?
Sweetwater Marsh Always tidal	California Department of Transportation	How do the wetlands that were constructed to mitigate damages to habitats of endangered species compare with the nearby natural wetlands? When will the constructed wetlands have complied with mitigation requirements?

season in spring; Coulter goldfields when in flower). The rationale for monitoring these various attributes and examples of data from these monitoring programs have been summarized elsewhere (PERL 1990). The question is, are there simple, low-cost indicators of wetland functions that might readily be used by nonexperts, or is this merely wishful thinking?

6.1.2 Indicators of Ecosystem Functioning

This section covers the development of simple indicators of ecosystem function, what might be used to test their validity, and what more may be needed to go beyond the single-species, single-habitat situation. Much more is yet to be done; no single indicator can ensure that habitats remain capable of supporting the biodiversity of the region's coastal wetlands.

Ability to support biodiversity. Southern California wetlands are valued for their biodiversity. A principal function that needs to be measured in both constructed and natural wetlands is the ability to sustain native species.

Species that make good indicators of a wetland's ability to support biodiversity have strong and specific links to one type of habitat. When that type of habitat is lost or damaged, the numbers decline. An argument is made that resident endangered species are good indicators for how well ecosystems function in sustaining biodiversity because they are the first to decline when their habitats are lost or damaged. Those endangered species that are animals and that feed high in the trophic web are also likely to be indicators of the condition of the whole ecosystem, though a contrary argument could

Table 6.2. Ecosystem Attributes Sampled Regularly at Four Southern California Coastal Wetlands

Attribute	San Elijo Lagoon	Los Peñasquitos Lagoon	Sweetwater Marsh Refuge	Tijuana Estuary
Date of first data	1989	1986	1986	1979
Water quality				
Levels of dissolved oxygen, temperature, salinity profiles	x	x	x	x
Nutrients in the water column				x
Fish				
Seines, or	x	x	x	x
Traps for exotic fishes			x	
Benthic invertebrates				
Core samples in channels	x	x	x	x
Marsh vegetation				
Occurrence and cover		x	x	x
Cordgrass heights and density			x	x
Cordgrass foliar nitrogen			sampled intermittently	
Coulter goldfields population		x		
Marsh soils				
Salinity		x	x	x
Nutrients			x	
Birds				
Belding's Savannah sparrows		x		

be made that specialist feeders are better indicators of the feeders' specific prey. For example, the wandering skipper *(Panoquina errans)* may be a good indicator of its salt grass food base, or vice versa, because the butterfly larvae feed only on salt grass.

The State of California recognizes 94 animal species as endangered or threatened with extinction (California Department of Fish and Game 1989). Three of these are bird species: the light-footed clapper rail, Belding's Savannah sparrow, and the California least tern. The first two are resident species; the tern is migratory. An additional endangered species is a hemiparasitic annual plant of the upper salt marsh: salt marsh bird's beak. Many wetland-dependent species are rare and threatened with extinction; these include insects (e.g., tiger beetles, genus *Cicindella;* wandering skipper). Still others are considered sensitive, such as the Coulter goldfields (*Lasthenia glabrata coulteri*).

Monitoring designed to assess a region's ability to support biodiversity should track the populations of sensitive species. Richard Zembal of the U.S. Fish and Wildlife Service monitors clapper rails throughout Southern California by using a network of volunteers. Counts are made during the highest tides of the year, when the birds move to the upper part of the marsh or are forced to become conspicuous by standing on floating debris.

Canopy architecture. Canopy architecture is used to assess the ability of vegetation in the lower and upper parts of the salt marsh to support endangered birds. Height distributions of cordgrass have been analyzed in relation to nesting of clapper rails, and performance standards have been developed (Zedler 1993; cf. section 4.1.3). Maximum heights and cover of glasswort are used to assess the status of high-tide refuges for clapper rails, and maximum heights of pickleweed are used to assess the abundance and distribution of perches for Belding's Savannah sparrows.

Tesingt how well the evaluation of canopy architecture might help in determining habitat to support the biodiversity of the whole marsh is a bigger project. It is not recommended that canopy architecture be used as a single indicator of nesting habitat for clapper rails because the correlation between tall cordgrass and nesting by rails does not indicate that short cordgrass is the only critical limiting factor. For this reason, PERL also monitors the abundances of forage items (crabs and other invertebrates) and distributions of habitat types (area and proximity of high-tide refuges).

Canopy architecture of cordgrass is likely to be useful in evaluating habitat for predatory beetles, whose presence is essential for controlling native scale insects *(Haliaspis spartina)* that can damage cordgrass (Boyer and Zedler 1996). The adult beetles are not aquatic animals. Hence, they require high-tide refuges, such as tall cordgrass stems (K. Williams, San Diego State University, personal communication). An outbreak of scale insects occurred at one constructed marsh island that lacked both tall cordgrass and beetles. No standards for height distributions have been developed for this critical carnivore, however.

Evaluations of canopy that can support nesting by Belding's Savannah sparrows have required a different approach, because pickleweed does not lend itself to measurements of height. Individual plants of this bushy, vegetatively reproducing perennial are hard to distinguish. Pickleweed also grows in a much wider range of habitats than cordgrass does, and it develops different growth forms under various degrees of tidal influence. Maximum heights and branch strength are important, though, because the birds tend to perch on and defend their territories from the highest plants available (White 1986; Powell 1993). We measure heights of pickleweed, but we have also experimented with remotely sensed variables that indicate physiological stress. In the section on remote sensing, Phinn and Stow (Section 6.4) describe the relationship between plant vigor and the normalized difference vegetation index (NDVI), which is obtained by measuring the reflectance of near infrared and red wavelengths. In our experimental mesocosms at Tijuana Estuary, growth of pickleweed (measured as branch elongation) was strongly correlated with the NDVI. This is a promising indicator of the vigor of pickleweed.

Other indicators. Cordgrass and pickleweed are but two of about two dozen vascular plant species that are important in Southern California salt marshes, and clapper rails and Belding's Savannah sparrows are only two of the thousands of species that the salt marsh supports. Thus, canopy architecture can be used to evaluate only one aspect of what is needed to maintain biodiversity. Beyond this initial effort, we need to determine how well the canopy information predicts habitat suitability for other species. Then, we need to work with additional plant/animal communities and species-specific dependencies.

Several measures of water quality can be used to assess potential for support of fishes and invertebrates. Indicators of stress include low concentrations of dissolved oxygen at the bottom of the water column early in the day, warm water temperatures, lowered water salinities, and high biomass of phytoplankton and macroalgae. These conditions sometimes develop at Los Peñasquitos Lagoon in late summer if it is closed to tidal action. These indicators lead us to recommend immediate dredging of the inlet to restore tidal flows and avoid a fish kill.

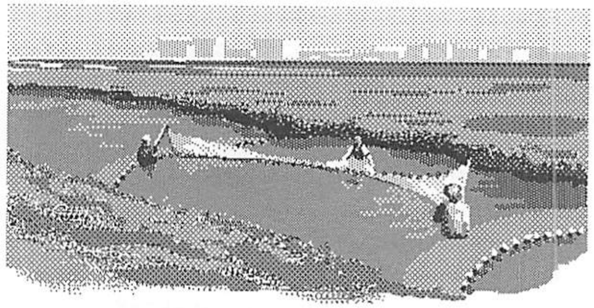
6.1.3 Testing Indicators

Suggesting an indicator is the first step in evaluating the functioning of constructed habitat. Acceptance by the scientific community is an important second step. If the indicator is adopted by peer reviewers, the third step is to work closely with agencies that manage endangered species to test the cause-effect relationship between the indicator and the functions it estimates. For height distributions of cordgrass, the variations in canopy architecture and water levels at the nest site during the nesting season of clapper rails need to be related to nest failure or success, and several marshes should be used as study sites. Because clapper rails are endangered, caution would be required for such testing. Remote recorders of water level would need to be installed near the nesting sites, and a means of assessing floatation and inundation of nests within plant canopies, without disturbing birds, would be required. If such tests supported the efficacy of the criteria, then managers would have greater confidence in the indicator as a measure of a suitable nesting habitat.

The use of cordgrass height distributions to indicate nesting by clapper rails grew out of our long-term (12-year) monitoring program at Tijuana Estuary, which led to an understanding of cordgrass dynamics and provided a rich data base for comparing well-used clapper rail habitats with unused constructed cordgrass marshes. The data from several cordgrass marshes at the Sweetwater Marsh National Wildlife Refuge, each of which has had a different history of use by clapper rails, made it possible to compare marshes with greater and lesser use by rails. Simultaneous study of clapper rail populations over many years and in several of the region's coastal wetlands were carried out by Zembal and Massey (1983); their insights into where nests are built and where and why nests fail were critical in making the link with specific canopy criteria. Finally, the availability of two areas that were constructed as habitat for light-footed clapper rails (but not yet used by rails) and planted with cordgrass at different dates (1984, 1985, and 1986), along with funding for study (1989 ff.), made it possible to compare canopy characteristics of unused habitats with those of natural, well-used sites for a period of years (Zedler 1993).

Simple systems with few species and few management issues may more readily use indicators. Certainly on the Atlantic Coast there is a long history of measuring cordgrass productivity to assess the importance of marshes to fisheries. In one case, the Great Sippewissett Marsh in Massachusetts, cordgrass productivity has been monitored for 21 consecutive years as a major response variable to wastewater addition (J. Teal, WHOI, personal communication). In Southern California, measurements of primary productivity were abandoned several years ago because they were both inadequate and too destructive.

A single indicator is unlikely to suffice in areas with a wide variety of species and where the principal management issue, that is, sustaining biodiversity, is so encompassing. However, the search for such indicators and criteria should continue.



6.2 IMPROVING METHODS OF ASSESSING FISH POPULATIONS

Thomas J. Kwak

Quantitatively sampling estuarine fish assemblages can be a difficult task because of the diversity of microhabitats that fish occupy within these ecosystems. A variety of fish-sampling gear has been used in salt marsh estuaries, including throw traps, drop nets, flumes, beam trawls, and seines. Of these, none is 100% efficient in collecting fish in any environment, and all have presumed and known biases that vary depending on sampling conditions, operator proficiency, and fish behavior. One means of reducing this variable bias and improving the accuracy of quantitative fish assessments is to estimate fish populations within assemblages directly by using an assortment of mark-recapture or removal methods (Ricker 1975, Seber 1982).

Tidal creeks and channels are the most prevalent fish habitats in most salt marsh estuaries in Southern California; some embayments are present, but to a lesser extent. The channel habitat of these estuaries can be further categorized into three generalized microhabitats: open-water, epibenthic, and benthic areas. Although fish freely move among these microhabitats, individual species are generally adapted to a specific environment. For example, the open-water microhabitat is most often occupied by topsmelt and California killifish, the epibenthic areas by flatfishes (e.g., California halibut and diamond turbot) and sculpins (e.g., staghorn sculpin), and the benthic areas by the burrowing gobiid species (primarily arrow goby). Obviously, no single fishing gear will provide a high sampling efficiency in all three microhabitats, but seining has been the favored technique for sampling adult and juvenile fishes in Southern California estuaries (PERL 1990).

In the study reported here, catchability and sampling error in population estimates were calculated from the September and December 1993 seine catch of fishes in Tijuana Estuary. The fish assemblage of Tijuana Estuary is sampled quarterly at three sites, as part of an ongoing, long-term monitoring project. Evaluation of the sampling gear (seine) provides a basis for refining sampling methods. The specific objectives were to (1) compare various measures of catch (as possible population indexes) with direct estimates of fish populations derived by using a maximum-likelihood removal method, (2) quantify seine catchability of several species of fish, and (3) assess the precision of mean weight estimates derived for individual species by measuring a subsample of fish vs. the entire catch.

6.2.1 Improving Quantitative Assessment

Fish were sampled in channel areas of Tijuana Estuary (150–200 m^2 each) enclosed by two 3-mm mesh block nets. After the measured sampling area was enclosed, fish were collected by using a 15.2-m-long, 2.2-m-wide bag seine, also constructed of 3-mm mesh netting. Seining was done from bank to bank, perpendicular to the flow of the water. Catch was recorded separately for each of three seine hauls; at sites where catch for any one species did not decrease with successive hauls, two additional seine hauls were collected (five total). All collected fish were measured for total length (± 1 mm) and weight (± 0.01 g). Large fish were measured in the field; small fish were preserved and measured later in the laboratory.

Population abundance, biomass, and catchability were estimated by using a removal method (maximum likelihood estimator; Seber 1982, Bohlin et al. 1989) separately for each species collected in the sampling area. Calculation of population estimates was facilitated by using a microcomputer software application specifically designed for this purpose (Kwak 1992). The removal population estimates were considered the most accurate estimate of the abundance of fish populations, and less rigorous measures of catch (single seine haul catch and sum of catch of three seine hauls) were compared with the removal estimates. This comparison allowed an assessment of the degree of underestimation that occurs if less rigorous measures of fish sampling are substituted as population measures or indexes.

Subsamples of 20 fish were randomly selected for weight measurements of each species at each site (if more than 20 individuals were collected). Mean weights calculated from the 20-fish subsamples were compared with the corresponding mean weights obtained by weighing all fish collected for that species and site. This comparison evaluated the degree of precision lost by weighing only a subsample of the catch vs. the entire catch.

6.2.2 Better Estimates Through Statistics

Less rigorous measures of fish catch were substantially lower than the corresponding removal population estimates. The catch from a single seine haul was, on the average, only 23.4% of the removal estimate, and the summed catch of three seine hauls was 51.3% of the estimate, but individual comparisons varied widely (Fig. 6.1). Thus, fish biologists who collect only a single seine haul in a sampling area are grossly underestimating populations and most likely are collecting only about one-quarter of the fish present. If three seine hauls are collected, about one-half of the fish present will be sampled. Therefore, it is important to express the results of fish collections obtained by using less rigorous sampling methods in terms of catch per unit effort or a species list rather than to erroneously present those data as estimates of population density or biomass. These results strongly support estimating populations directly, by using either a removal or a mark-recapture method, in assessments of fish assemblages. Furthermore, for all quantitative sampling, an enclosed area should be used to prevent escape of fish during sampling and to add an area component so that population estimates can be converted to density (e.g., number of fish per square meter).

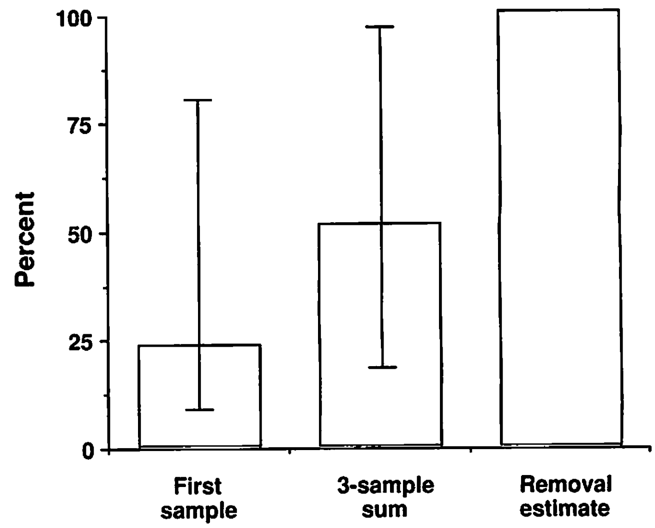


Figure 6.1 Comparison of less rigorous sampling catches (first seine haul sample and sum of three seine haul samples) to the corresponding removal population estimate for each species and site. Data are mean (± range, $N = 6$) percent comparison of less rigorous catch to the removal estimate, assuming that the removal estimate is the best representation (100%) of the population.

Average catchability for the three most common species of fish in Tijuana Estuary (topsmelt, arrow goby, and California killifish) was between 0.40 and 0.50 of a maximum possible value of 1.0 (Table 6.3). Mean catchability of the staghorn sculpin was higher (0.55), and that for diamond turbot was lower (0.12). Fish that were rarely captured had catchabilities of either 1.00 or zero; that is, rare fish are either all caught (catchability = 1.00) or all escape (catchability = 0). The wide variation in catchability within and among species further supports using statistical methods to estimate populations. The relative abundances of species in the seine catch may be more indicative of their catchability when a seine is used than of their actual occurrence in the sampling area. Population estimates calculated separately for each species will correct for such variable gear bias.

The mean absolute difference between population mean weights calculated on the basis of a 20-fish subsample vs. weighing all fish collected was 9.6%.

Table 6.3. Catchabilities of Fish Collected in Tijuana Estuary in September and December 1993

Species	No. of Sites Sampled	Mean	SE	Range
Topsmelt (*Atherinops affinis*)	2	0.45	0.27	0.18–0.73
California killifish (*Fundulus parvipinnus*)	3	0.41	0.18	0.07–0.65
Arrow goby (*Clevelandia ios*)	3	0.46	0.18	0–1.00
Staghorn sculpin (*Leptocottus armatus*)	3	0.55	0.22	0.32–1.00
Diamond turbot (*Hypsopsetta guttulata*)	2	0.12	0.12	0–0.24
Gray smoothhound (*Mustelus californicus*)	1	1.00		
Longjaw mudsucker (*Gillichthys mirabilis*)	1	1.00		
Opaleye (*Girella nigricans*)	1	1.00		
Bay pipefish (*Syngnathus leptorhynchus*)	1	1.00		

Note: Maximum catchability = 1.00.

Considering the effort that can be reduced when fish catch for an individual species may be more than 1,000 individuals, a less than 10% loss in precision can be justified for estimating fish biomass by measuring a subsample of 20 individuals.

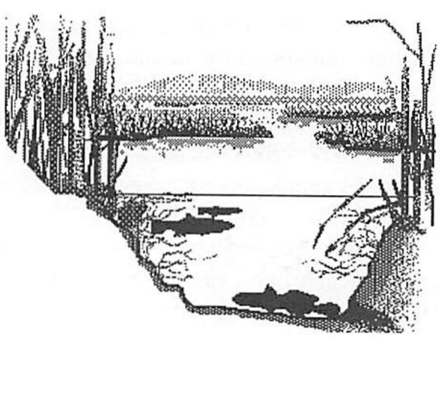
6.2.3 A Proposed Standard for Estuarine Populations

A three-sample removal population estimate (maximum likelihood method), in which a bag seine is used in an area confined by block nets, and lengths and weights of 20 individuals per species are measured, is proposed as a practical standard to quantify estuarine fish populations in Southern California. The continuation of this study will increase sample sizes and may include additional species to add more certainty to the interpretation of these findings.

6.3 ANALYSIS OF THE FOOD WEB AND FISH-SUPPORT FUNCTIONS

Thomas J. Kwak

Two basic questions about the functioning of wetland ecosystems are which primary producers provide the bulk of the organic matter that fuels the food web, and whether grazer- or detrital-based food chains are more prevalent. Ecologists have begun to examine ratios of naturally occurring stable isotopes of carbon, nitrogen, and sulfur to determine sources and sinks of materials (Peterson and Fry 1987). Studies of stable isotopes in food webs can indicate sources of organic matter that feed consumers and can provide information on trophic relationships within ecosystems. Measurements of stable isotopes are particularly useful in studies of estuarine food webs because these systems are thought to be based on detritus, and the origin of detrital material is difficult to determine by other means.



Animals and their diets are similar in isotopic composition for carbon, sulfur, and nitrogen. Isotopic ratios (see formulas in next section) for these three elements change slightly but predictably when the elements are assimilated (Peterson and Fry 1987). Thus, unique "signatures" are formed by combinations of multiple stable isotope ratios, and these signatures can generally be traced through the food web with some slight change among trophic levels. In this manner, the ultimate source of food support for various consumers (especially ecologically important fish and birds) can be determined if the signatures of potential sources are sufficiently distinct. Information on the trophic level of consumers can also be ascertained by examining the degree of fractionation in nitrogen isotope ratios among consumers.

6.3.1 Technical Background

It may be necessary to contract with an outside laboratory to do stable isotope analyses because of the expense required to set up and maintain the required instruments. Although no published list is available, a number of laboratories in the United States and Canada specialize in stable isotopes and will do analyses for a fee. Ratios of stable isotopes of carbon, nitrogen, and sulfur in organic materials collected in Southern California salt marshes were determined by Coastal Science Laboratories in Austin, Texas. Each sample was completely converted to a gas by combustion and separated into pure gases (CO_2, N_2, and SO_2). Each pure gas was analyzed by using an isotope ratio mass spectrometer, and the isotopic composition was quantified relative to a standard reference material. Standards were carbon in the PeeDee limestone, nitrogen gas in air, and sulfur from the Cañyon Diablo meteorite. Results for each element were expressed as parts per thousand differences from the corresponding standard (∂):

$$\partial X = [(R_{sample}/R_{standard}) - 1] \times 10^3,$$

where X is ^{13}C, ^{15}N, or ^{34}S, and R is the corresponding ratio of $^{13}C/^{12}C$, $^{15}N/^{14}N$, or $^{34}S/^{32}S$. The ∂ values include a measure of both heavy and light isotopes, whereby increases in ∂ denote relative increases in the amount of heavy isotope.

6.3.2 Role of Detritus in Atlantic Coast Wetlands

Early studies of stable isotopes in tidal creeks of a Georgia salt marsh showed that the detritus in that system was isotopically more similar to phytoplankton than to cordgrass (Haines 1977), and subsequent research suggested that consumers were being supported, at least in part, by phytoplankton (Haines and Montague 1979). These findings brought into question the long-standing dogma of a vascular plant source of detritus that supported salt marsh food webs. Investigators have since reported various relative influences of vascular plants and algae on the salt marsh food web (e.g., Hughes and Sherr 1983, Jackson et al. 1986, Sullivan and Moncreiff 1990).

Most of these ecosystem-level studies have been done in Atlantic coastal marshes; no similar studies have been done in Southern California salt marshes. Relative to Atlantic salt marshes, Southern California estuaries are distinct in topography, hydrology, vegetation, and fauna and in anthropogenic influences on the environment and its biota (Zedler 1982). Thus, the relative importance of vascular plants and algae as an ultimate food source for invertebrate, fish, and bird consumers remains unknown for Southern California salt marshes. The finding (Zedler 1980) that algal mats can be as productive as the vascular plant overstory in a Southern California salt marsh casts further doubt that the dogma of a vascular plant, detritus-based food web applies to these ecosystems. Given the great amount of effort and cost allocated in attempts to restore and enhance estuarine habitat in Southern California, the origin of the food web in these systems is vital information that will be required to understand ecosystem functioning and guide restoration efforts.

6.3.3 Role of Southern California Salt Marshes in Supporting Fish

Ongoing studies at PERL are the first comprehensive attempt to determine the source of foods for consumers in Southern California coastal marshes. The four primary goals of the research are to (1) determine the relative importance of vascular plants and algae as the base of the estuarine food web that supports higher trophic levels, including fish and birds; (2) assess the influence of sewage inflows on the food web; (3) determine principal foods of estuarine fishes; and (4) gain insight on trophic relationships among consumers in salt marshes. By combining stable isotope studies with direct assessment of fish food habits, both ultimate and direct fish-support functions can be determined. Successful achievement of these objectives will provide information useful in managing, restoring, and understanding the ecology of Southern California salt marshes.

Data and organic materials were collected in March-April and August-September 1994 from four sites within the northern arm of Tijuana Estuary and from two sites in San Dieguito Lagoon. Although both salt marshes have been altered and degraded by human activities, Tijuana Estuary remains less disturbed and more ecologically functional than San Dieguito Lagoon. Comparing findings from a fully tidal, functioning system with findings from one more degraded will add insight to be applied in restoration planning. Sewage-derived organic matter was collected directly from a sewage collection facility in the city of Tijuana and from a canyon on the southern edge of Tijuana Estuary that receives raw effluent regularly.

Samples analyzed for stable-isotope composition included both human influences (sewage-derived organic matter) and salt marsh primary producers as potential sources of the organic matter that forms the base of the estuarine food web. Primary producers were collected from tidal creeks and both high- and low-marsh habitats and included four genera of macroalgae, two genera of cyanobacteria, and four species of salt marsh vascular plants. Detritus in the water column was assessed directly as a critical link in the food web that makes immediately available the energy and nutrients from the primary producers (or sewage) to the consumers. Sampling included 14 invertebrate, 7 fish, and 2 wetland resident bird species to represent primary and secondary consumers (Table 6.4). Although all species contribute ecologically to the estuarine food web, several have more obvious value to humans. Two fishes of commercial and recreational importance, California halibut and diamond turbot, and the endangered bird, the light-footed clapper rail, were included in the study.

Selected findings from this study (Kwak and Zedler, in review) include the following:

- Isotope ratios of producer groups were statistically differentiated most clearly by the carbon isotope, then by sulfur, and least clearly by nitrogen. Considering the three isotopes collectively further distinguished the producers. This was an important first step in clarifying the base of the food web.

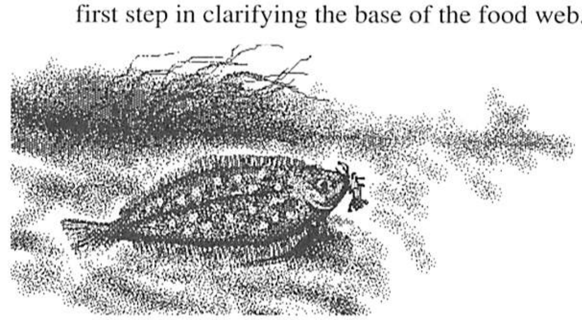


Table 6.4. Summary of Organic Materials Collected from Tijuana Estuary and San Dieguito Lagoon for Determining Multiple Stable Isotope Ratios of Carbon, Nitrogen, and Sulfur

Material	No. of Taxa	No. of Sites	No. of Samples
Tijuana Estuary			
Particulate organic matter	NA	4	4
Sewage-derived organic matter	NA	2	4
Cyanobacteria	2	2	2
Macroalgae	4	4	11
Vascular plants	4	4	20
Invertebrates	14	3	14
Fish	7	3	16
Birds	2	NA	4
Total	33	6	75
San Dieguito Lagoon			
Particulate organic matter	NA	2	2
Macroalgae	3	3	5
Vascular plants	1	2	2
Fishes	3	2	9
Total	7	3	18
All Locations			
Particulate organic matter	NA	6	6
Sewage-derived organic matter	NA	2	4
Cyanobacteria	2	2	2
Macroalgae	4	7	16
Vascular plants	4	6	22
Invertebrates	14	3	14
Fishes	7	5	25
Birds	2	NA	4
Total	33	9	93

- The distribution of isotope ratios of fish in San Dieguito Lagoon was significantly more tightly clustered (less variance) than that of fish in Tijuana Estuary, suggesting that a more complex food web exists at Tijuana Estuary.
- Fish isotopic signatures were significantly different between wetlands for carbon, but not for nitrogen or sulfur, which may be related to the relative scarcity of C_4 plants at San Dieguito Lagoon.
- Results of a cluster analysis indicated natural groupings of isotope ratios of broad producer and consumer groups that appear to be related to habitat type and salt marsh elevation.
- Trophic enrichment of the nitrogen isotope suggested that at least four trophic levels exist in the Tijuana Estuary food web and that the light-footed clapper rail was the highest-level consumer among those sampled.

In general, our results indicate that the food web of Southern California salt marshes is more complex than those of Atlantic and Gulf of Mexico coastal sites where similar analyses have been done. The isotopic composition of major consumers relative to that of producers suggests that macroalgae, cyanobacteria, and vascular plants (*Spartina*) all contribute to the food-web base. It also appears that sewage inflows to Tijuana Estuary provide a minimal contribution, if any, to the estuarine food web.

Relative contributions of producers and sewage to the food web in these systems cannot be defined precisely because of the number of potential sources (i.e., quantitative mixing models cannot be developed). This difficulty reveals a limitation in applying stable isotope techniques to food-web studies of salt marsh ecosystems that may be particular to Southern California systems. Southern California coastal marshes are confined to narrow stream outlets along the coastline that are usually surrounded by steep, rugged topography (Zedler 1982). This physiographic setting is in marked contrast to the broad, flat coastal plains of the Atlantic and Gulf coasts that form expansive estuarine systems. Intertidal salt marshes along the Atlantic and Gulf coasts may encompass many kilometers and typically support vast, nearly monospecific vegetation (e.g., Hackney and Haines 1980), whereas in Southern California systems, such as those we studied, high-and low-marsh habitats are much more spatially proximate and may be separated only by meters.

The spatial proximity of habitats and associated vegetation in Southern California marshes likely influenced the finding of multiple sources contributing to a complex food web. Thus, more potential sources of organic matter should be considered in food-web studies of these systems, relative to similar studies conducted on the Atlantic and Gulf coasts. As more potential sources are added to analyses, however, it becomes increasingly difficult to distinguish patterns among producers and between producers and consumers. Incorporating additional isotopes into a study could improve the ability to make distinctions, but the point remains that the technique has limitations in complex ecosystems.

Our findings have important implications for wetland management. Because no single habitat in the estuarine ecosystem can be excluded as supporting producers that contribute to the food web, all intertidal habitats (tidal creeks, salt marshes, pools, salt pans) should be protected, restored, and managed when the objective is to develop or enhance fish or bird habitat in Southern California coastal marshes. We anticipate that our results will be used in future negotiations about mitigation plans for areas designed to replace habitat that supported fish. It is reasonable to conclude that salt marsh habitats contribute to fish support; hence, restoration of both channel and salt marsh habitat could be considered mitigation for lost fish habitat.

6.4 USE OF REMOTE SENSING TO MONITOR PROPERTIES OF VEGETATION

Stuart Phinn and Douglas Stow

Remote sensing of terrestrial vegetation is based on the principle that the amount of electromagnetic radiation reflected or emitted from plants is determined by the type of plant and the plant's biophysical characteristics (e.g., height, horizontal and vertical leaf areas, biomass, chemical composition, and productivity). There are differences in the amount of electromagnetic radiation from the sun reflected in different wavelengths from wetland plants and soils (Fig. 6.2). Remotely sensed data (e.g., aerial photographs, satellite images, data collected with hand-held radiometers) have been used extensively to delineate wetland areas and to map the composition of their communities. These data are also useful for measuring and mapping the biophysical characteristics of wetland areas. Remote sensing has several advantages over field sampling approaches. It is nonintrusive, so measurements can be made in inaccessible areas, and it provides repeatable coverage on micro to regional scales. Remote sensing does not replace field sampling, but it does provide synoptic coverage to guide ecological work.

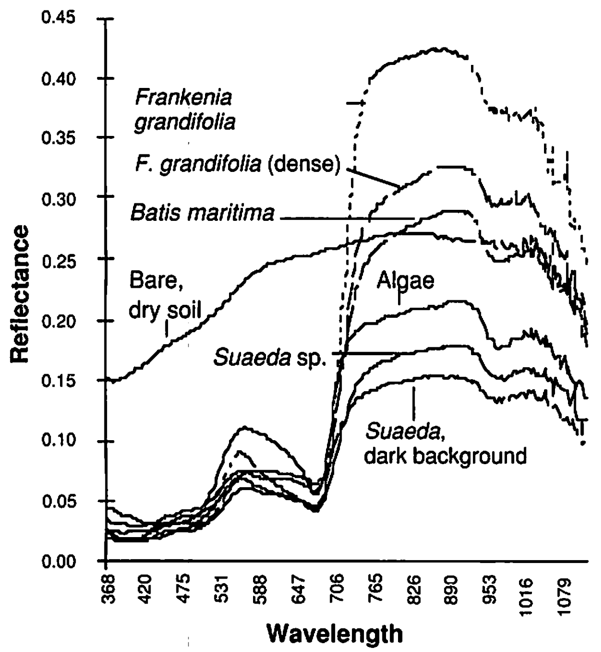


Figure 6.2 Representative samples of reflectance spectra from salt marsh habitats using a hand-held spectrometer.

Historically, the development of remote-sensing applications for mapping and monitoring coastal wetlands has centered on the extensive wetlands of the Gulf and Atlantic coasts. Summaries of research relating remotely sensed data to characteristics of wetland vegetation have been published (Carter 1978, Hardisky et al. 1986, and Gross et al. 1989).

The tools and techniques used in wetland applications (field radiometers, multi-spectral scanners, and digital videography) are explained here for monitoring vegetation in restored coastal wetlands. Most research on use of remote sensing in wetlands has focused on establishing and testing relationships between the spectral reflectance and biophysical characteristics of wetland vegetation for relatively large areas of monotypic, undisturbed wetlands. In our experience and in our review of the literature, we have not found any applications of multispectral, digital remote sensing for monitoring restored wetlands. Also, little information is available on remote sensing of the smaller, species-rich, and highly disturbed Pacific Coast wetlands. Monitoring disturbance and restoration in these areas requires information on the composition, structure, and other characteristics of vegetation at specific spatial and temporal scales (Kusler and Kentula 1989). Remote-sensing techniques can be used to provide these data, as shown by studies on the East Coast. Developing and using these techniques and data should improve ecological monitoring and adaptive management in wetlands.

In the sections that follow, we focus on three potential applications of remote-sensing techniques for wetlands monitoring restoration of wetlands:

1. Determining and mapping the areal extent of individual species and assemblages
2. Assessing the structure of the vegetation or landscape (horizontal and vertical)
3. Assessing the condition of the vegetation

Several case studies will illustrate our experience with these techniques in Southern California.

6.4.1 Applications

Use of remote-sensing techniques helps establish ecological indicators of the type and condition of vegetation in wetland ecosystems (Halvorson 1991, Loehle 1991). Previous use of remote sensing in monitoring East and Gulf coast wetlands has been at local to regional spatial scales (1–1,000 km^2) in monotypic, relatively undisturbed wetlands. In contrast, applications for wetlands restoration and monitoring in Southern California require data on micro to local scales (<1–100 ha) because the wetlands are small (often fragmented) and heterogeneous in species composition. Techniques developed on the East and Gulf coasts had to be refined before they could be used on the West Coast.

Before remote sensing is used to characterize wetlands, careful consideration should be given to the spatial and temporal scales being examined as well as the information required to make management decisions. Suitable data and techniques for analysis are then selected. It is necessary to determine if the data required satisfy the three basic types of information at a suitable scale to be provided by remotely sensed data (Table 6.5, columns 1 and 2).

The rest of this section outlines available data and techniques to address each of the three basic information categories.

6.4.2 Mapping the Areal Extent of Species and Assemblages

Spatial and spectral resolution. At micro to local scales (1 m^2 to 1 km^2), remotely sensed data can be used to

Table 6.5. Information Requirements and Suitable Data and Analysis Techniques for Monitoring Wetlands Restoration

Information Required	Specific Features	Spatial (Temporal) Scales	Data	Analysis Techniques
Areal extent and location of marsh vegetation species	Wetland boundaries Vegetation species composition Habitat composition	Micro to regional (1–10 years)	Data from airborne video or scanner systems Scanned color infrared photographs High-resolution satellite images Ancillary digital data (e.g., topography)	Manual delineation If patch > pixel size → per-pixel classifier + ancillary data → knowledge-based classifier If patch < pixel size → mixture model approach
Vegetation patch characteristics (landscape structure)	Type Size, shape, complexity, connectivity, dispersion Edges Diversity	Micro to local (1–10 years)	Unprocessed images Calibrated and classified images Existing wetland boundary or vegetation maps	Exploratory spatial data analysis Landscape structure metrics
Vegetation structure and condition	Individual plant/canopy horizontal and vertical dimensions Biomass Net primary productivity Processes (photosynthesis, transpiration) Stress Trace gas flux	Micro to regional (month–10 years)	Spectral radiance measurements (narrow or broad band) Corrected images Biophysical parameters associated with spectra or images	Field radiometry and spectroscopy Establish relationships between spectral indexes and biophysical parameters Scale up radiometer relationships to pixels

delineate patches on the ground representing the extent of a particular marsh type, a characteristic assemblage of species, an area of stressed plants, or the extent of a wetland. In order to determine the appropriate type of data and analytic technique for delimiting these patches, several factors must be considered. These include the size of the region of interest (extent), the characteristic scale of the patches of vegetation (pure stands or wetland boundaries), temporal variability of these patches, and the type of information required (boundary location, areas, biophysical characteristics).

For delimiting wetland boundaries and mapping the areal extent and distribution of various wetland plant species, generally speaking, the spatial resolution of the data should be no greater than half the size of the smallest feature to be defined. For example, if the minimum width of stands of cordgrass is 2.0 m, an image with picture elements (pixels) less than 1.0 m would usually be necessary to delimit these areas. Similarly, the spectral properties (specific wavelengths of sensed electromagnetic radiation) of the data influence the amount of information that can be extracted about the condition and structure of the vegetation. Some wavelengths are more sensitive to absorption by pigments in vegetation (controlled by leaf chemistry), whereas others are more sensitive to scattering by leaf structure and moisture content. Spectral resolution increases in the following order of types of remote sensing data: color infrared aerial photographs, multispectral images (<20 bands), and imaging spectrometer data (>100 bands). The greater number of bands potentially allows more information on scene characteristics to be established.

Infrared and color infrared aerial photography at scales from 1:1,000 to 1:10,000 have been used successfully to delineate wetland boundaries and to determine the extent and condition of various species of marsh vegetation (Anderson and Wobber 1973). These techniques are relatively cost-efficient and are outlined in a number of introductory texts on remote sensing.

In contrast, multispectral or hyperspectral digital image data store multiple spectral brightness values per pixel, a format amenable to computer processing. An image can be considered a number of grids, and each grid layer contains brightness values for a specific waveband. The brightness value in each pixel is a function of the number of different vegetation types in the pixel and their relative abundance, structure, and condition. Brightness values can be converted to physical quantities such as spectral radiance or spectral reflectance when suitable calibration data exist.

Image classification. Automated processing of image data with commercially available software packages (ERDAS, PCI, ER-MAPPER) makes it possible to detect groupings of pixels with similar brightness values in all bands. This process is referred to as image classification and uses multivariate statistical clustering algorithms to delimit groups of pixels with similar values in each spectral band. The analyst determines the number of classes and suitable ranges of pixel values in each image band and often provides "training data" with statistics on the required classes. These groupings are assumed to represent patches of a specific type or condition of vegetation. Field verification is also required to check this. Either the raw image data or special combinations of image bands can be used in image classification. Specific transforms or combinations of bands such as the NDVI and principal components analyses produce pixel values that are more highly related to the characteristics of the structure (biomass) and condition of vegetation than the raw bands are (Jackson 1986). The NDVI is an index that ranges from −1 to +1. "Healthy," dense vegetation has high NDVI values, and stressed or dead vegetation usually has values near or below 0.

For applications in which the area of interest (e.g., patches consisting of uniform assemblages of vegetation) is larger than the pixel size, per-pixel classification techniques are used. These algorithms use multivariate clustering routines to detect groups of pixels with similar spectral brightness values. These clusters or spectral classes are assumed to represent areas of distinct types of vegetative cover on the ground. If ancillary data are available for the restoration site, these can be combined with the remotely sensed data in a modified classification routine. For example, if information on the topography and hydrology are in digital format and compatible (as raster or vector-based files) with the imagery, they can be used to help in the classification. The spectral brightness values for each pixel are assessed to determine the vegetation class the pixel is likely to fall within, and this allocation is checked against the elevation range and distance from channel for that class. Most image-processing systems have the capability to implement these schemes with combined geographic information system modules.

For applications in which the area of interest is smaller than the pixel size, subpixel analysis techniques are used (mixture models). Each pixel's spectral brightness values are assumed to result from a certain combination of reflectance values produced by each object and background in the pixel. If the spectral reflectance values for each different type of object are known from previous hand-held radiometric or spectrometric work, the percentage of each pixel covered by each object type can be estimated. Useful explanations of the mixture-modeling approach have been published (Ustin et al. 1986, Adams et al. 1993).

6.4.3 Assessing the Structure of Vegetation or Landscape

The purpose of examining the structure of the landscape in restored wetlands is usually to describe (preferably quantify) the horizontal and vertical variation in assemblages of various plant species. Once the characteristic structures are established, they can be related to or used as ecological indicators for the status of the wetland ecosystem. Information is provided for evaluating the condition or functional capability of the wetland by quantifying the structure of patches of vegetation in wetlands (e.g., detecting patches of specific types of vegetation and determining their areal extent, density, shape, and connectedness to other types of vegetation). Depending on the type of information required, two approaches can be used with remotely sensed data to establish descriptions of landscape structure.

First, the most common form and size of patches of vegetation, characteristic length scales of landscape features, or the overall degree of spatial variance in types of land cover types can be determined by exploratory spatial data analysis. These techniques can be applied to individual spectral band images or to transformed images of airborne multispectral imagery. In order to establish descriptive information on landscape structure, such as the range of commonly occurring patch sizes, spatial statistical measures such as variance windows of different sizes, semi-variograms, correlograms, and other structure functions can be applied. These measures have primarily been used with imagery in nonwetland areas (Woodcock et al. 1988a, 1988b; Simmons et al. 1992) and have been discussed extensively in terms of their potential application to ecological patterns (Turner and Gardner 1991, Rossi et al. 1992, Simmons et al. 1992).

The second approach generates more specific information on landscape structure. "Landscape metrics" can be applied to classified images (see previous section) or to digitized maps of types of wetland land cover. Landscape metrics are indexes that summarize the spatial characteristics for one particular patch or for a collection of patches of one type of vegetation. These metrics, which are often used by researchers in landscape ecology, can provide insight into the processes or conditions that produce a particular landscape pattern (O'Neill et al. 1988, Turner 1989, Turner and Gardner 1991). Several patch characteristics can be quantified by using FRAGSTATS (McGarigal and Marks 1994), a public-domain software package (Table 6.6). This and similar programs require input in either pixel- (grid) or vector-based (polygonal) representations of maps of vegetation type. In both cases, remotely sensed data can be classified to delimit polygons of vegetation type that can then be analyzed by using landscape metrics. Each landscape metric should be fully understood before it is applied to data to make an interpretation. For example, does it represent the composition or the configuration of the landscape, or both? What aspect of configuration? Which scale is spatially explicit? These questions and others should be asked to determine if the landscape metric represents the landscape structure in an ecologically meaningful way and can be used to assess restoration. The metric should provide information on

Table 6.6. Landscape Metrics Computed In FRAGSTATS

Scale	Metric
Area metrics	Area Landscape similarity index Class area Percentage of landscape Total landscape area Largest patch index
Patch density, size, and variability metrics	Number of patches Patch density Mean patch size Patch size standard deviation and coefficient of variation
Shape metrics	Shape index Fractal dimension Landscape shape index Area-weighted mean shape index Double log and mean patch fractal dimension Area-weighted mean patch fractal dimension
Core area metrics	Core area, number of core areas Core area index, percentage of landscape Core area density Mean core area per patch Patch core area standard deviation and coefficient of variation Mean area per disjunct core Disjunct core area standard deviation and coefficient of variation Total and mean core area indices
Nearest neighbor metrics	Nearest neighbor distance Proximity index Mean proximity index Nearest neighbor standard deviation and coefficient of variation
Diversity, contagion, and interspersion metrics	Shannon's and Simpson's diversity indexes Patch richness Patch richness density Relative patch richness Shannon's and Simpson's evenness index Modified Simpson's diversity and evenness index Interspersion and juxtaposition index Contagion index

Source: McGarigal and Marks 1994.

vegetation or habitat conditions (an ecological indicator) that are related to the restoration goals and performance criteria.

6.4.4 Assessing the Condition of Vegetation

Most work in which remote-sensing techniques were used to assess the condition of wetland vegetation were based on plot-scale (0.1–1.0 m^2) relationships between hand-held spectral radiometer or spectrometer measurements and simultaneous measurements of biophysical properties (e.g., aboveground biomass, stem lengths, and leaf area indexes). Two approaches are commonly used, empirically derived relationships and deterministically based models. Both approaches involve two stages. First, the properties of the vegetation (structure, condition, radiation transfer) that control its spectral reflectance properties are determined. Second, a deterministic model or empirically derived relationship is used to relate one or several of the measured biophysical parameters of the vegetation to spectral measurements (e.g., Asrar 1989, Gross et al. 1989). The model or relationship can be inverted and the measured spectral reflectance values used to infer biophysical parameters for the vegetation at that site.

Hand-held radiometry. The instruments used most often to obtain ground measurements of spectra of wetland vegetation are spectral radiometers (4–8 broad wavebands) or field spectrometers (>100 narrow bands). Descriptions of these instruments' operation, applications, and different models have been published (Milton 1987). Radiometers and spectrometers record the strength of radiation reflected from surface features or incident from the sun and atmosphere from within an area determined by the instrument's field of view. Filters are used to limit the range of wavelengths sensed. A spectrometer senses in more and narrower ranges of wavelengths than a radiometer does.

Once the spectral bands and required biophysical parameters have been determined, a suitable spatial and temporal sampling scheme can be designed. Spatially, the sampling scheme should be within pure stands of a specific community of vegetation for which a relationship or model is being developed and should represent the range of structural and spectral variability in the community. The number, spacing, and size of sample plots should be selected according to the spatial scale of the vegetation structures in each type of community (Curran and Williamson 1986, Webster et al. 1989). Further stratification is often required within communities because of structural differences that affect their spectral reflectance characteristics (e.g., short, medium, and tall plots for cordgrass). Consideration of the community's phenology and growth form will influence the temporal dimension of the sampling program. Two approaches are suggested: sampling at times characteristic of each stage in the vegetation's growth cycle (e.g., spring green-up, summer greenness peak, fall senescence, and winter dormancy) and sampling at the period of peak greenness to represent vegetation condition. Careful records should also be made of the sky and environmental conditions at each sampling date to account for any variability in data not explained by the measured biophysical parameters.

Once large samples of spectral reflectance and biophysical parameters have been collected at one or several periods over the growing season of the vegetation of interest, relationships can be developed and models applied. The strength of the relationships or sensitivity of the model to the range of measured values should be examined first. The established relationship or model outputs are then validated. For the empirical approach, either simple regression or correlation techniques can be used; the biophysical parameter is the dependent variable, and the measurement of reflectance is the independent variable. In deterministic models, the measurements of spectral reflectance are transformed and used in equations to estimate the variable of interest. Sensitivity and error analyses can be done with these models to determine the effects of errors in the input. Measured biophysical parameters are used to validate the output of the model.

Imagery. Two approaches are commonly used to relate spectral brightness values from an image of a restoration site to the condition of vegetation on the ground. Both approaches are empirical. The first uses a relationship already established (e.g., for radiometric data) between spectral reflectance and a biophysical parameter. In the second, this relationship is established between pixel values and conditions of ground vegetation sampled over larger areal units to correspond to the large ground resolution element of an image (compared with a handheld radiometer). Spectral brightness values are extracted from each pixel corresponding to the field sampling sites. The image data should be calibrated to absolute reflectance or radiance values made at the time of the overflight.

6.4.5 Case Studies

In San Diego County, Caltrans is currently responsible for creating two intertidal wetlands along the eastern shore of San Diego Bay. The following sections summarize how remotely sensed data were applied to provide information for monitoring the progress and success of various Caltrans restoration projects.

Sweetwater Marsh restoration project: Habitat for light-footed clapper rails. Construction of the Highway 54 overpass with Interstate 5 in Chula Vista in 1984 damaged a part of the tidal wetlands in the Paradise Creek Marsh and filled areas adjacent to Sweetwater Marsh. One objective of mitigation was to provide sustainable vegetated home ranges for the light-footed clapper rail. These birds require a habitat with a specific mixture of low-, middle-, and high-marsh areas. Within the low part of the marsh, cordgrass must also provide suitable cover of a specified height. In order to assess the success of efforts to provide mitigation, the number and areal extent of potential home ranges (Fig. 6.3A) of clapper rails must be assessed, and areas of vegetation of low-, middle-, and high-marsh species must be calculated. We derived these areal estimates by using high-resolution airborne digital multispectral image data.

An airborne data acquisition and registration (ADAR) System 5500 image acquired in June 1995 was used in the image-classification procedure outlined here. Results of a previous image-classification exercise in Tijuana Estuary showed that the different assemblages of plant species were most separable on this type of imagery during the March–April or July–August period (Stow et al. 1993). Specific details for the image (Fig. 6.3b) are outlined in Table 6.7.

In order to delimit patches composed of similar marsh vegetation, a per-pixel supervised classification was applied to the data. On the basis of initial field inspections, areas of known types of ground cover were selected as "training sites" for specific assemblages of vegetation. Those areas where training sites could not be defined because of mixtures of vegetation (e.g., high marsh sites) were classified by using computer-generated classes. Once the image was classified, the accuracy of class labels and boundaries was field checked. This check ensured that each class was labeled according to the dominant type of marsh vegetation occurring in its boundaries and that the boundaries did delimit zones of transition. Mislabeled classes were then corrected, and poorly defined classes were reclassified. Labels for the final image classes were selected to enable direct evaluation of the vegetation distribution as potential habitat for clapper rails and for separation into high-, middle-, and low-marsh zones.

The final product from the classification and field checking was a classified image of the Sweetwater Marsh restoration site on June 10, 1995 (Fig. 6.3B). Pixels within the same marsh type (high, middle, or low) were given the same label and merged (e.g., all those cover types considered to represent assemblages of high-marsh vegetation were aggregated into one class). Results are summarized in Table 6.7, indicating which potential home ranges exhibit the required acreage under current and future conditions. Seven home ranges for clapper rails are close to meeting the criteria, and three could comply with future expansion of marsh vegetation. The estimates indicate an increase in areas of low-marsh type, above the required threshold for each potential home range. The site was reflown in 1995, and areas of dense cordgrass and high marsh were not the same for the two census periods. This was because of differences in water levels and illumination, both of which affect the spectral separability of plants.

Sweetwater Marsh: Monitoring the effects of soil amendments. Because restored marsh soils are deficient in nutrients and nitrogen, the effect of fertilizing them is being established (Boyer and Zedler 1996, and in preparation). Hand-held radiometric measurements were obtained for a variety of pure cordgrass plots treated with different amounts of nutrients (urea added in different months and from 2 to 12 times). The spectral reflectance data were examined in relation to cordgrass total stem length, density, and amount of foliar nitrogen to aid ecological monitoring.

An Exotech four-band radiometer was used to measure spectral reflectance in blue, green, red and near-infrared wavebands of the treated plots. A single measurement was obtained over the center of each of 70 2-m × 2-m plots of pure cordgrass. There were four transects on the North Connector Islands and three transects on the South Connector Islands. Each transect contained 10 plots. Within each transect, nutrient treatments were assigned by using a randomized block design. Measurements were taken at three stages in the 1993 growing season, in April, July, and September.

For each sampling date, analysis of variance (ANOVA) was used to determine if there were significant differences between the NDVI values for plots that received different treatments. Regression analyses were then done to assess possible relationships between the vegetation indexes (e.g., NDVI) and measurements of stem length. A temporal-differencing analysis was also done for all the data collection periods to determine whether there were significantly different spectral index changes in plots given different treatments.

Statistically significant differences were observed for NDVI values associated with different treatments (Fig. 6.4). As the number of applications of nutrients increased,

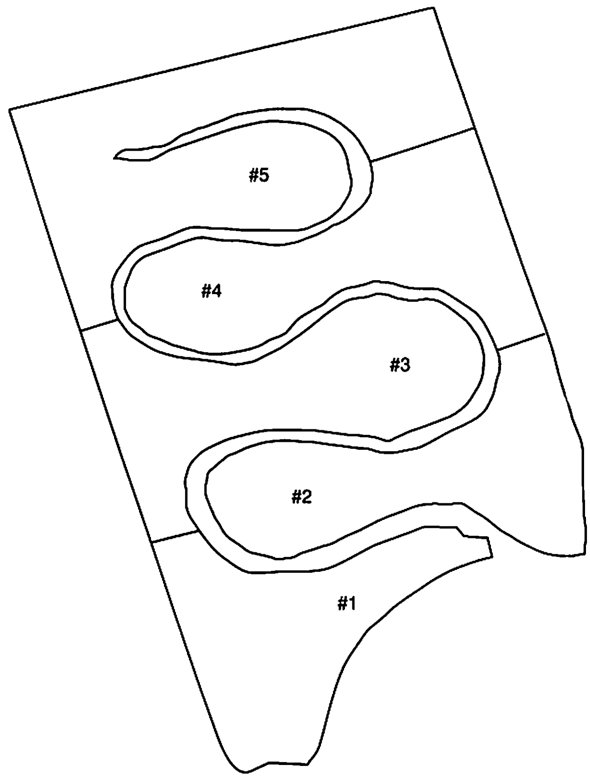


Figure 6.3A Potential home ranges for the light-footed clapper rail at a 6.9-ha constructed marsh within Sweetwater Marsh National Wildlife Refuge. Polygons of suitable size (0.8-1.6 ha) were drawn to include low, middle and high marsh habitats. The area of each habitat type within each potential home range was then assessed using ADAR imagery (Fig. 6.3B).

so did the NDVI values. This suggests that the NDVI is sensitive to changes in cordgrass structure and abundance that were observed when the radiometric measurements were obtained. For the regression relationships between cordgrass spectral and biophysical measurements, no statistically significant r^2 values were observed for NDVI, near-infrared, or red reflectance values regressed against corresponding stem length, height, density, and foliar nitrogen values in all plots. Several factors may account for the low r^2 values: (1) Senescing vegetation had long stem lengths but low NDVI values; (2) Standing water in some plots reduced NDVI values; (3) Plots with flattened stems had higher NDVI values than plots with stems of the same length that were not flattened; and (4) Spectral measurements were insensitive beyond a particular level of biomass. Further work is required to determine the exact plant characteristics represented by the spectral reflectance data for single sample dates and over time.

Sweetwater Marsh: Cordgrass spectral and biophysical parameters. Total stem length of cordgrass can be inferred from spectral vegetation indexes because both quantities relate specifically to plant biomass (Gross et al. 1989). Hand-held radiometry provides an efficient and

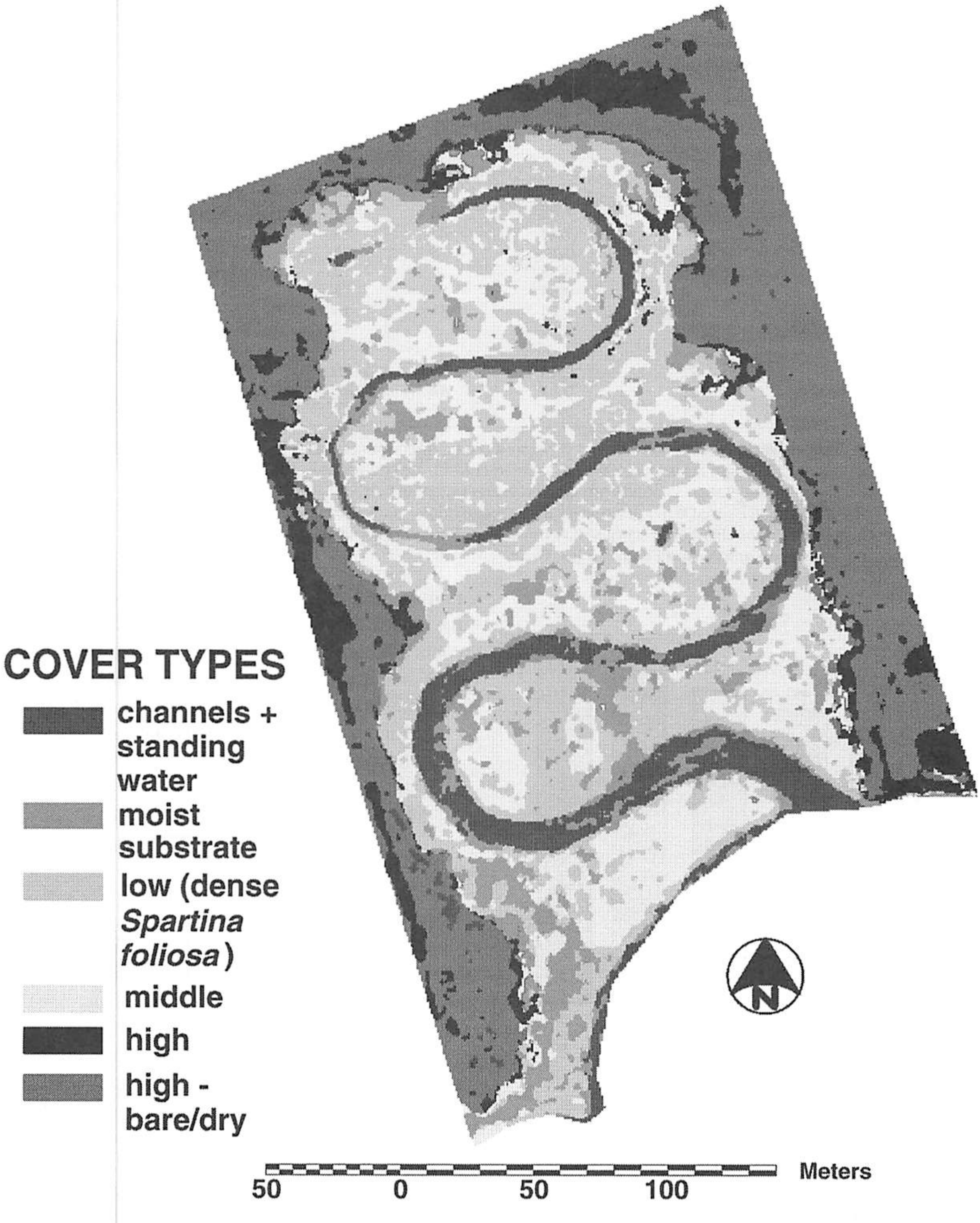


Figure 6.3B Marisma de Nación, Sweetwater Marsh National Wildlife Refuge, showing habitat types differentiated from an airborne data acquisition and registration System 5500 image taken on June 10, 1995. Black-and-white prints do not distinguish all cover classes, but low, middle, and high marsh habitats are clearly indicated on the color version, which is used for management purposes.

Table 6.7. Potential Home Ranges for the Light-Footed Clapper Rail

Location	Polygon Area (hectares)	Marsh Area (hectares)	Habitat Type (%) High	Habitat Type (%) Low
Compliance criteria	0.8–1.6	≥0.8	≥15	≥15
Current				
North Connector islands 1–4	1.07	0.74	27	13
South Connector islands 1–2	2.28	1.58	38	10
South Connector islands 3–5	1.84	1.44	23	44
South Connector island 6	1.05	0.71	13	15
Marisma area 3	1.23	0.73	16	13
Marisma area 4	1.24	0.84	19	28
Marisma area 5	1.90	1.24	31	16
Future				
North Connector island 5	1.92	0.74	13	30
Marisma area 5 (north sector)	1.02	0.61	47	40
Marisma area 5 (south sector)	0.91	0.64	15	81

Note: Polygons that approximated the criteria (0.8- to 1.6-hectare areas, each with at least 15% low marsh and at least 15% high marsh) in 1994 are labeled current, and polygons that might meet the criteria as vegetation expands are labeled future potential home ranges. The term potential indicates that birds are not yet using these home ranges and that other essential attributes of usable home ranges may be lacking. Polygons contain marsh, channel, and some bare areas. See Figures 6.3A, B.

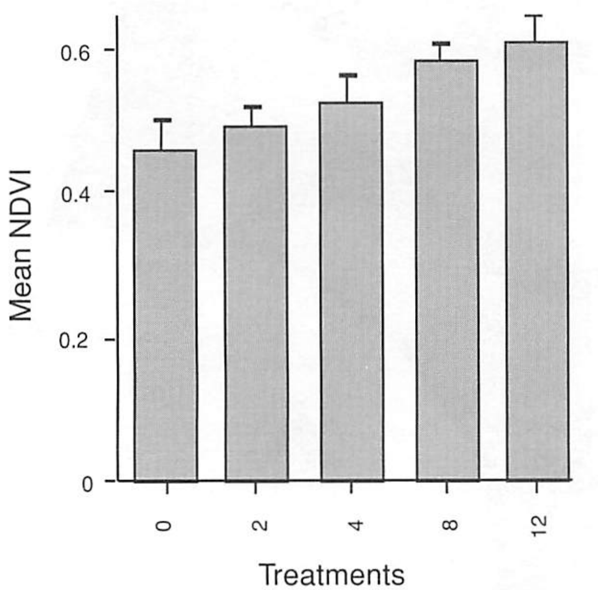


Figure 6.4 Mean values of the normalized difference vegetation index (NDVI) for cordgrass vegetation in experimental plots at Sweetwater Marsh National Wildlife Refuge. Treatments differed in the number of times (0-12) that nitrogen was added; N=7.

noninvasive means of establishing the total stem length in cordgrass plots and guides more detailed analyses by ecologists. Similarly, hand-held radiometry may provide an efficient means to monitor other changes in cordgrass structure and condition.

A conceptual model (Fig. 6.5) has been developed from our research on the variations of spectral vegetation indexes for various cordgrass canopy characteristics and from previous work on the controls of vegetation spectral reflectance characteristics. On the basis of the model, the following factors should be taken into account as contributing to the strength and weakness of the relationships between NDVI values and cordgrass biomass (stem length, height, density):

- Canopy architecture (percentage of leaf surface in horizontal vs. vertical planes, light penetration through canopy)
- Surrounding features that influence spectral values (algal coverage, soil moisture, degree of water coverage)
- Time of day, solar elevation
- Proportion of dead biomass and browning of canopy
- Possible existence of asymptote in the relationship between vegetation indexes and green biomass (i.e., increases in green biomass do not increase the vegetation index beyond a certain level)

Tijuana Estuary: Mesocosm experiment with pickleweed. The effect of different tidal conditions on the restoration of pickleweed was examined at the PERL Mesocosm Facility at Tijuana Estuary (Ross 1994, Callaway et al., in preparation). Three tidal treatments (full tidal, tides excluded, and tide waters impounded) were assigned to 24 1-m × 10-m mesocosms according to a randomized block design. The mesocosms were planted with pickleweed. Hand-held radiometry was used to explore whether the measured spectral reflectance values could provide useful information for monitoring the growth of this plant.

Measurements were made at monthly intervals for light transmission, growth (elongation of tagged stems), bare space, and plant cover (binary absence or presence of pickleweed along transect lines). Detailed measurements of plant cover were made down to 1.0-cm intervals

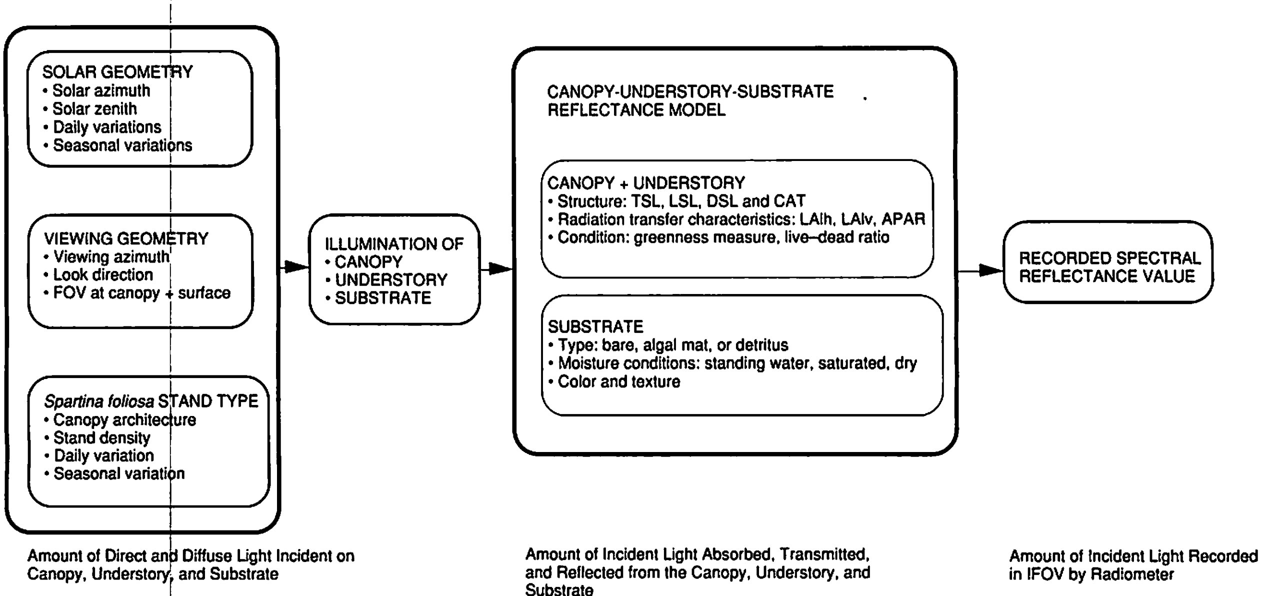


Figure 6.5 *Spartina foliosa* spectral reflectance model V.1.

in each 10-m^2 mesocosm. Within each 1-m section of each mesocosm, three radiometric measurements were obtained with an Exotech radiometer with a circular "footprint" 30 cm in diameter. Spectral radiance was also measured over halon-coated calibration panels and used to convert radiance measures to spectral reflectance factors. Measurement dates in 1993 were in April, June, and August.

The NDVI was used to assess relationships between the recorded radiance values and pickleweed cover for each type of treatment. ANOVA was used to determine if there were significant differences between NDVI values for different tidal treatments. Interval mean cover index and bare index were examined in relation to spectral reflectance data for each sampling period. A temporal-differencing analysis was also done by subtracting NDVI, near-infrared, and red values recorded in June from corresponding values in April and values in August from those in June. ANOVA of these difference values was used to determine whether different treatments caused significantly different changes in spectral index.

Statistically significant correlations were observed between NDVI values and the following variables (in descending order of correlation): bare space, pickleweed cover, and light transmission for data collected during April, June, and August. Significant differences were observed in NDVI and other spectral measurements between the different treatments (full tidal, inundated, and excluded). In most cases, individual mesocosms had corresponding positive NDVI/cover and negative NDVI/bare space correlation values. Analysis of measured differences in cover (as a surrogate for plant growth) for each trench in relation to changes in recorded NDVI values over the same periods also indicated a significant positive correlation.

According to the results obtained, spectral data from hand-held radiometric measurements appear to be sensitive to variations in pickleweed cover in the mesocosms over time (Fig. 6.6). With the exception of several outliers (produced by standing water in areas of medium cover), increases in NDVI values corresponded to measured increases in pickleweed cover (Fig. 6.6a). A correspondingly strong negative relationship was evident between NDVI and bare space (Fig. 6.6b). These results indicate that the NDVI is reflecting increases in the amount of pickleweed cover over time (Fig. 6.6c). In fact, the radiometer was able to detect stressed pickleweed much more readily than were researchers measuring cover and height of vegetation. The only measurements that showed evidence of pickleweed stress were the growth data, which require marking and remeasuring pickleweed branches. Such work is far more labor intensive than hand-held radiometry. Hence, remote-sensing techniques offer potential for rapid assessment of vegetation stress. Further analysis is necessary to account for the effects of the type and state of the soil background (moist, dry, salty) and the change in pigmentation in pickleweed over the growing season.

6.4.6 Summary

Monitoring wetland restoration requires information on the composition, structure, and other biophysical characteristics of vegetation at specific spatial and temporal scales. Remote-sensing techniques can provide such data for the characteristically small and fragmented natural and restored wetlands in Southern California. Three specific applications of remote-sensing techniques for monitoring wetlands restoration are (1) identifying and mapping the areal extent of different species and their assemblages, (2) assessing vegetation or landscape

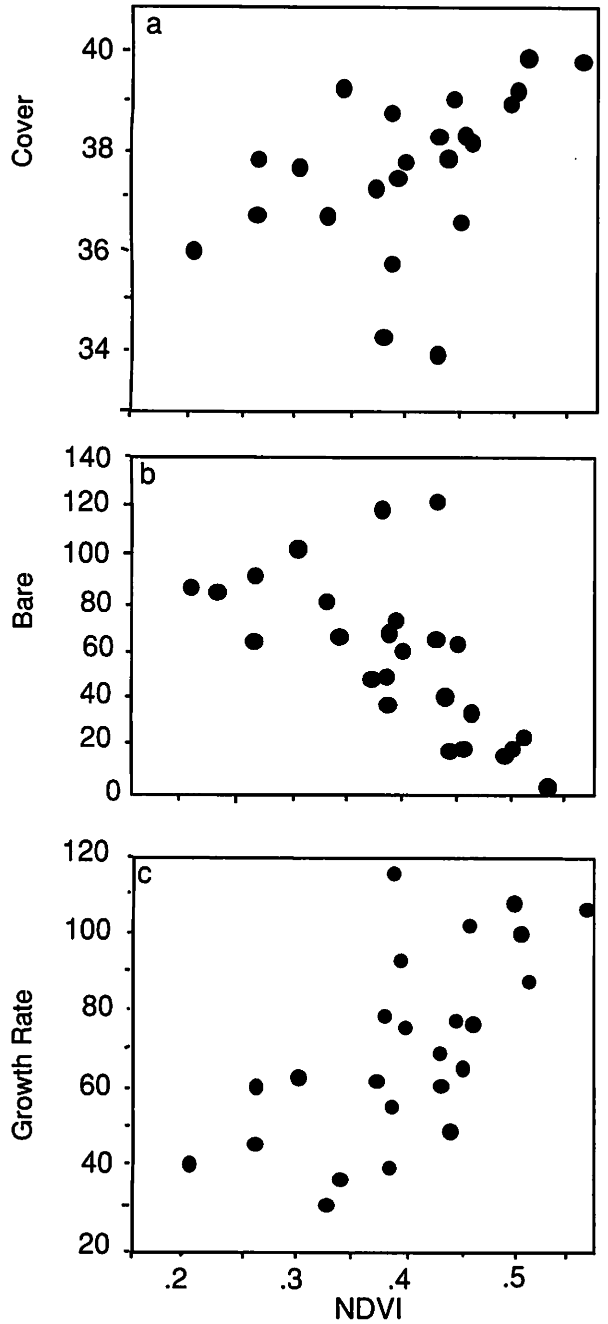


Figure 6.6 Scatter plots of NDVI and vegetation cover, bare space, and growth rates of pickleweed grown in mesocosms at Tijuana Estuary.

structure (horizontal and vertical), and (3) assessing the condition of vegetation.

Remote-sensing techniques are suited to monitoring restoration for several reasons: (1) their noninvasive and nondestructive nature, variable scales of coverage, and capability of repeated acquisitions; (2) their ability to be quantitatively related to structural and conditional properties of vegetation; and (3) their cost efficiency.

6.5 ADAPTIVE MONITORING AT TIJUANA ESTUARY: Assessing Interannual Variability

It is a paradox that an estuary in one of the mildest climates in the United States ranks among the most environmentally and biologically variable. Throughout the year, the average monthly temperature ranges only from 9° to 21°C (48° to 69°F, in San Diego, NOAA data), and average daily solar insolation ranges from 178 to 606 langleys (Chula Vista, 1976–1977 data). Frost is extremely rare along the coast, and snow is virtually unknown. However, whereas temperature is predictably moderate, rainfall and streamflow are not. Neither the amount of freshwater influent nor its timing is dependable.

Tijuana Estuary is simultaneously controlled by streamflow volumes and tidal circulation, both of which are subject to catastrophic change. Within the past decade, environmental conditions have ranged from prolonged drought during an 8-month nontidal period to prolonged freshwater inflows.

Natural climatic variations are compounded by land management actions (for example, flood control, reservoir discharges, wastewater discharges, and coastal land uses that foster erosion). The combination of extreme events and human disturbance has caused major swings in estuarine structure and functioning, with incomplete recovery in between. The story of Tijuana Estuary is one of high interannual variability in climate, sea level, streamflow, and tidal flushing.

Management of Tijuana Estuary is a shared responsibility. Both the California Department of Parks and Recreation and the U.S. Fish and Wildlife Service are concerned with maintaining habitat that will support several of the region's endangered species. Although quantitative studies of the vegetation began in 1974 (Zedler 1977), and sampling of a large number of fixed stations began in 1979 (Zedler 1983; Zedler et al. 1986; Zedler, Nordby, and Kus 1992), the site did not become a National Estuarine Research Reserve (NERR) until 1982. In 1984, the monitoring became a part of research funded by the NERR program, and in 1988, the monitoring program was formalized by NOAA. At various times, we reevaluated the measures, sites, and frequency of sampling to document and understand interannual variability of the salt marsh.

In the sections that follow, background data on the physiography and climate are summarized to explain why the salt marsh has variable environmental conditions (especially soil salinity). The dynamics of the vegetation are summarized, and the changes in the monitoring program are explained. Finally, the value of an adaptive approach to monitoring is discussed.

6.5.1 Physiography and Climate

Geographic setting. Tijuana Estuary (lat. 32°34'N, long. 117°7'W) is the southwesternmost estuary of the continental United States. Its location on the tectonically active Pacific Coast indicates a dynamic geologic setting. This drowned-river-type estuary is small (1024 ha) relative to river mouths along the more gentle topography of the Atlantic and Gulf of Mexico coasts.

The 448,323-ha watershed is mostly (73%) within Mexico. One large reservoir, behind Rodriguez Dam on Tijuana River, and two small U.S. reservoirs on Cottonwood Creek, together control 78% of the watershed upstream of Tijuana Estuary. Situated 8 km upstream of the estuary is the metropolis of Tijuana, Baja California, a growing city of about 2 million inhabitants. Between Tijuana and the estuary is an urbanized and agricultural river floodplain with no pristine habitat.

Rainfall patterns. Southern California has a dry Mediterranean-type climate, with mild, wet winters and warm, dry summers. Averages (cf. section 1.1) suggest that rainfall increases gradually from October to a peak in January and February; this is followed by a gradual decline through April. The average annual total is 25.2 cm. Averages are misleading, however, because a single month may include much of the year's total rainfall. In December 1921, rainfall was 24 cm, nearly equal to the annual average. The month with the most variable rainfall is December (SE = 0.4); the least variable is June (SE = 0.2).

Rainfall data are better characterized by modes and measures of variation than by means. For 134 years, the modal year had less than 25 cm of rain, and only 22 years were within 10% of the average. The coefficient of variability for annual rainfall is 41%, the SE is 0.9, and the range is 8–70 cm. Because of these high interannual variations in the timing of rainfall and in total rainfall, the potential for variation in streamflow is great.

Streamflow. Natural patterns of streamflow to Tijuana Estuary no doubt changed greatly with the construction of the first dam in 1912. Since then, flows have been impounded within reservoirs, and peak velocities have been dampened. With the diversion of Colorado River flows to Southern California and northern Baja California, imported water has been introduced to the Tijuana River, mostly in the form of renegade sewage flows from the City of Tijuana (cf. section 3.2) In recent years (about 1986–1991), Tijuana River had greater summer flows because of persistent flows of raw sewage from Mexico (ca. 13 million gal/day). However, these flows are dwarfed by variations in winter flood flows and reservoir discharges. Interannual variability is high for Tijuana River (Fig. 6.7). As with rainfall data, annual streamflows are better characterized by modes and extremes than by means.

Variability within months is also high; SEs range from 0.8 to 230.5 cubic meters. The least variable months are

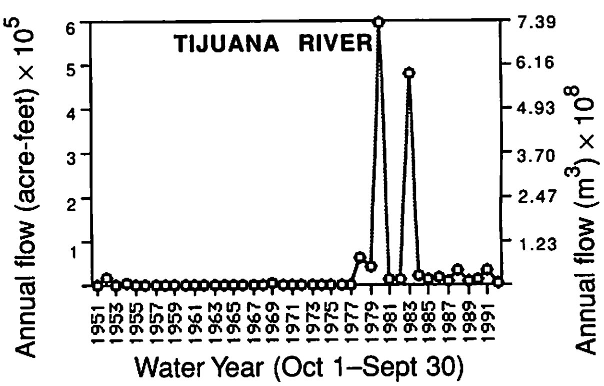


Figure 6.7 Streamflow in the Tijuana River shows many years with no flow, followed by flooding in 1978, 1980, and 1983. Data are from the International Boundary and Water Commission's gauge at the United States-Mexico border, about 8 km upstream of Tijuana Estuary.

those with low streamflow. Seasonal flows lead to high intra-annual variability; years of low rainfall have little winter streamflow and little difference in streamflow from month to month. This is particularly true for series of dry years, when runoff is accommodated by the upstream reservoirs (cumulative capacity about 160 cubic meters). However, reservoirs can also prolong streamflows, and continuous discharge can reduce seasonality and month-to-month variability. During 1983, record flows occurred from March through December when Rodriguez Reservoir was lowered to restore its flood storage capacity.

Tidal flushing and sea level dynamics. Tijuana Estuary is influenced by semidiurnal mixed tides. Approximately 50% of the tidal prism (the average volume of water exchanged by the tides) is attributable to the northern arm (Oneonta Slough) of the estuary (Williams and Swanson 1987). Historic air photos show that whereas the main channel has been affected by wave washovers and dune erosion, the smaller channels and tidal creeks have changed little (Zedler, Nordby, and Kus 1992).

The tidal prism was increased when a landlocked lagoon was connected by a dredged channel sometime before 1924. This lagoon was used as a sewage oxidation pond and made tidal to discharge through the estuary and into the ocean until the 1960s. Otherwise, the tidal prism has consistently declined. Williams and Swanson (1987) calculated changes from topographic maps from 1852 (rough drawing of open water, marsh, and upland), 1976 (1.5-m contours), and 1986 (30-cm contours). Their results (about 2 million m^3 in 1852 to 0.4 million m^3 in 1986) show an 80% loss in tidal prism over the 134-year period. Seven major flood events occurred between 1852 and 1986, filling the central embayment and the river and tidal channels. The 1980 flood dumped about 2 m of sediment along the southeastern margin of the estuary. In 1986, 50% of the tidal prism occurred in the northern arm, which accounts for less than 25% of the reserve. Elsewhere in Southern California, Mugu Lagoon lost 40% of its low-tide volume after consecutive floods in

1978 and 1980 (Onuf 1987). Sedimentation events are rare but are substantial when they occur.

Mean sea level rises about 21 cm/century along the San Diego shore (Flick and Cayan 1985). However, during the 1983 El Niño, mean sea level was 15 cm higher than predicted (R. Flick, personal communication). In January 1983, waves washed over the barrier dune at Tijuana Estuary and moved large quantities of sand into the main northern tidal channel. Sedimentation from the barrier dune reduced the tidal prism and led to closure of the ocean inlet. By early April 1984, tidal scouring was severely diminished, and the mouth of Tijuana Estuary became blocked with sand deposited by the longshore currents. The inlet remained closed until December 1984, when a major dredging program was completed and the estuary and ocean were reconnected.

History was repeated in January 1988, when a second major storm washed sand into approximately 80 m of tidal channel adjacent to the southern dune. This reduced tidal flows to the southern arm of the estuary (Fink 1989). The January 1988 storm was of much shorter duration than the 1983 storm, but it occurred during high tide (Flick and Badan-Dangon 1989).

6.5.2 Adaptive Monitoring

The monitoring program at Tijuana Estuary was neither planned nor executed with a long-term focus. Perhaps because there were no major investments in planning and no contracts to fund specific data gathering, we were free to improve sampling from time to time. The ability to tailor sampling is now recognized as appropriate for monitoring a highly variable system with changing management needs.

Sampling of 102 stations began in 1979 and was continued annually through 1988. The intent of sampling in 1979 was to characterize the cordgrass community across its intertidal range, from pure stands along the tidal creeks to about MHHW. The initial stations were established along eight transects perpendicular to tidal creeks, with 0.25-m^2 quadrats marked at 5-m intervals. Soil salinity was sampled at each station by expressing water from a 10-cm soil core onto a refractometer. Elevations of each station were measured on a separate date, which necessitated marking each station. Cordgrass was sampled by measuring the height of each stem in the 0.25-m^2 quadrat. Measurements were recorded for each quadrant of each quadrat, a step that made it possible to compare smaller and larger sampling areas. Pickleweed and other marsh plants are less readily measured because of their sprawling growth form. Estimates of the percentage of cover in six classes (0, <1, 1–5, 6–25, 26–75, 75–100) were therefore used to quantify the abundance of pickleweed.

Because stations were marked, it was possible to resample them in 1980, after major flooding. Once a major change in cordgrass was documented, it was possible to determine how long the change persisted. Hence, sampling continued in 1981 and 1982. Where cordgrass density was unusually high, we subsampled the 0.25-m^2 quadrats by randomly selecting one quadrant per quadrat. Estimates of pickleweed cover were inadvertently omitted in 1982, although occurrence was recorded. Also during 1982, Tijuana Estuary became part of the NERR system. This maintained our interest in the site and, when the 1983 flood occurred, we repeated sampling to assess a second freshwater event.

After 5 years, the investment in annual sampling seemed worthwhile (cf. results reported in Zedler 1983, Zedler 1986, Zedler et al. 1986), and funding was sought. Beginning in 1984, NOAA supported the sampling effort as part of the NERR program. The nontidal period coincided with the receipt of funding, and monitoring stations were expanded (to 216) to assess broader effects of the system-wide drought. Three of the initial transects were extended farther inland, and two new transects were added. We sought to sample more of the estuary but were constrained by topography in areas where transects abutted disturbed areas.

In 1988, after 10 years of annual sampling, we reviewed the entire monitoring program again. Several shortcomings were acknowledged, and improvements were recommended. Although the expanded sampling program had added 114 stations, most (85) were still within the range of elevations of the initial 102 stations sampled. Areas of flatter topography were oversampled (155 stations between 71 and 110 cm National Geodetic Vertical Datum, NGVD), and lower slopes (49 stations between 51 and 70 cm NGVD) and upper slopes (41 stations between 111 and 141 cm NGVD) were undersampled. Soil salinities were oversampled (little variation among stations of similar elevation) for the top 10 cm, but the salinity of deeper soils was ignored. Sampling was also time consuming, because the widespread transects and sampling stations were hard to locate. Finally, although estimates of cover were adequate for assessing major changes in canopies, they were too crude (±25%) to reveal lesser patterns.

Management issues for the region's coastal marshes were also considered. Three data needs were clear: (1) Reference data were needed for cordgrass marsh, pickleweed marsh, and upper-marsh habitats, so that success criteria could be established for mitigation projects;

(2) At Tijuana Estuary, a plan to restore tidal flushing to the southern arm had been developed, and before-and-after data were needed; (3) A major question at Tijuana Estuary was how sewage flows from Tijuana were affecting the salt marsh.

To meet these needs, we chose to sample representative areas of the marsh more intensively than before and to compare results from areas in both arms of the estuary. The sewage question also suggested a sampling scheme that was based on elevation (because sewage would reach lower intertidal areas more often than higher areas) and proximity to the sewage-laden river. We placed sampling stations near the mouth of the river and northward to the most inland tidal creeks. We placed one station near the first area to be restored. To retain our investment in stations with a decade of accumulated data, we retained the option to resample them at intervals (e.g., after a catastrophic event occurs).

Annual sampling became more intensive, but the number of stations was reduced. To equalize effort per marsh type and to improve sampling precision for mid- and upper-marsh communities on the basis of the dominant vegetation, we selected four transects at each of three marsh elevations. Areas of pure cordgrass along tidal creeks were selected to represent lower marsh. Areas dominated by pickleweed were selected for middle marsh, and areas with glasswort and shoregrass were chosen as upper marsh. A 20-m transect was staked at all 12 locations (three elevations, four sites each). Soil salinity sampling was reduced to three replicates per transect, but depths of 5–10 cm and 25–30 cm were sampled. In order to increase sampling precision and provide measures that were readily comparable between species, cover data were taken along 20-m transects, measuring intercept to the nearest 10 cm. Cordgrass heights continued to be measured in quadrats, with smaller size (0.10m^2) becoming the standard. The number of quadrats was set at five per transect.

6.5.3 Environmental Change Within the Estuary

The changes in rainfall, streamflow, and tidal flushing have led to high interannual variability in estuarine water salinity, intertidal soil salinity, and soil moisture. No long-term records of water salinity are available. However, the estuary is usually marine except during major winter streamflows. The nontidal period of 1984 was exceptional. After closure, the estuary water became brackish because of impounded river water, and then hypersaline because of evaporation during the long summer drought. By late summer 1984, channel water was several degrees warmer than normal, with about 60 ppt salt and higher phytoplankton populations than ever before measured (Fong 1986; Rudnicki 1986; Zedler, Nordby, and Kus 1992).

In most years, the salinity of marsh soils (measured in the top 10 cm) decreases during winter rains and streamflows and increases throughout summer (PERL 1990). Highest values are reached in September, when tidal amplitudes are low. The greatest interannual variations coincided with closure and drought (1984), extreme flooding (1980), and prolonged reservoir discharge (1983).

There is also considerable spatial variability in soil salinity; the highest intertidal elevations have the greatest range and seasonality (Zedler 1982). Within the lower intertidal part of the marsh, daily tidal inundation moderates the change in salinity, and soils are generally hypersaline (about 40–45 ppt).

Soil moisture varies spatially along with salinity. During September, drought and low tidal amplitude reduce surface soil moisture in the upper part of the marsh. A similar pattern may also develop in April if there is no rainfall, because tidal amplitude is lowest in spring. At this time of year, the lowest tides occur during the daytime, and soils in the upper part of the marsh can dry out. The 8-month nontidal period of 1984 included little rainfall, and soils in the marsh and tidal creek became dry and cracked, with white salt crusts marking former pools (Zedler, Nordby, and Kus 1992).

Effects on salt marsh vegetation. Tijuana Estuary is largely salt marsh habitat. The northern arm has the largest expanse of lower intertidal marsh; the southern arm is dominated by upper marsh and habitat that is transitional to the upland. Pacific cordgrass and perennial pickleweed *(Salicornia virginica)* dominate the lower marsh, whereas a variety of halophytes intermix in the upper part of the marsh (Zedler, Nordby, and Kus 1992).

Interannual variations in salinity and soil moisture affect both the composition and growth of marsh plants. Nontidal conditions produced the greatest changes to Tijuana Estuary, with several species declining in abundance and distribution. Some mortality was not apparent until 1985. Cordgrass was eliminated from more than half of the 102 lower-marsh monitoring stations. At the same time, the perennial pickleweed expanded its distribution from 75% to 87% of the sampling stations. Data from 215 intertidal monitoring stations show additional vegetation dynamics. An annual pickleweed (*S. bigelovii*) was nearly eliminated from the Oneonta Slough marsh. A short-lived succulent called sea-blite declined to very low frequency. These changes are related to low soil moisture, which differentially affects shallow-rooted annuals and deep-rooted perennials. The perennial pickleweed is known to produce deep roots that can follow a declining water table (Griswold 1988).

Lesser variations in plant growth have followed flooding and reservoir discharges. Growth of cordgrass and pickleweed changed with soil salinity in the following pattern (Zedler 1983, Zedler et al. 1986). Periods of low streamflow (higher salinity, shorter submergence times) favored pickleweed, whereas heavy streamflow (with low salinity and prolonged inundation) favored cordgrass. Cordgrass grew best with prolonged reservoir discharge, and plants channeled excess photosynthate into vegetative reproduction, so that both height and stem density increased substantially. These effects became clear from the monitoring record and experimental work of Griswold (1988).

Associated, confounding factors. The interannual variations in Tijuana Estuary biota are not entirely due to salinity and moisture. Several factors complicate the responses of vegetation to soil conditions. Nutrient inputs accompany streamflow, especially those resulting from sewage spills. The marsh is known to be nitrogen limited (Winfield 1980, Covin 1984, Fong 1986, Rudnicki 1986, Covin and Zedler 1988), and some of the growth stimulus associated with reduced salinity is related to increased input of nitrogen.

Competitive interactions among the plants influence their respective abundances; best known is the interaction between cordgrass and perennial pickleweed (Zedler 1982, Covin 1984, Covin and Zedler 1988, Griswold 1988). Covin and Zedler (1988) showed experimentally that pickleweed is the superior competitor for nitrogen. Thus, streamflows with low nitrogen influx may produce different responses than wastewater discharges.

Animal abundances both affect and are affected by the marsh vegetation. Where nitrogen is present in excess, plants concentrate it in their leaves and provide more nutritious foods for phytophagous insects. Covin (1984) suggested that outbreaks of a dipteran (*Incertella* sp.) were responsible for high mortality of cordgrass in fertilized plots at Tijuana Estuary. One endangered species, the light-footed clapper rail, depends on the lower-marsh vegetation and intertidal conditions. The number of rails dropped to zero or near-zero during the 1984 nontidal drought (P. Jorgensen, Tijuana River National Estuarine Research Reserve, personal communication). Reduced cordgrass cover, lack of invertebrate foods, and increased accessibility to terrestrial predators no doubt interacted to cause the rails' temporary demise.

6.5.4 Recovery and Long-Term Implications

The effects of rare, extreme events are persistent. The ecosystem has limited resilience, at least when faced with multiple catastrophes within 5 years. The shift of annual pickleweed from local dominance to rare occurrence is probably permanent. The habitat where it formerly occurred is greatly altered by the presence of a dense canopy of perennial pickleweed. The seed bank of the annual was apparently lost, when most of its seeds germinated in spring 1984 and died before reaching reproductive age (Zedler, unpublished data).

Cordgrass was slow to recover its predrought distribution. Its frequency of occurrence in the 102 monitoring stations ranged from 86% to 93% before tidal closure (1979–1983). It gradually expanded from a low of 38% in September 1985 to 50% in 1986 and 75% in 1988. Recovery most likely has been slowed by its chief competitor, perennial pickleweed (Griswold 1988). Cordgrass recovery was faster at the lower elevations, where tidal inundation is frequent and soils are anaerobic.

The population of light-footed clapper rails had increased to approximately 16 pairs by 1988, but it was not until 1991 that the population equaled its predrought density (41 nesting pairs; Zembal 1991). The recovery period was 7 years. If the existing population has built up from only 1 or 2 pairs, rather than by immigration, the rail's ability to persist in the long term may now be restricted by low genetic diversity. The species has some ability to disperse to upstream or to other coastal habitats (Zembal et al. 1985), but intramarsh movements are not common. Thus, lowered genetic diversity and long-term effects seem likely.

Regional implications. High interannual variability in Southern California coastal bodies of water produces highly unstable populations of intertidal salt marsh organisms. The effects that have been documented at Tijuana Estuary are mirrored in the region's other 25 coastal wetlands. Those that remain open to tidal flushing and occasionally have winter floods (but not prolonged heavy streamflows from reservoirs or wastewater sources) support the highest numbers of native species (cf. Table 2.4). Of 19 vascular plant species that were selected for regional comparison (PERL 1990), 18 are found in Tijuana Estuary, and 16–19 occurred in five other tidal ecosystems of Southern California (Sweetwater Marsh, Mugu Lagoon, Anaheim Bay, Upper Newport Bay, and Mission Bay Reserve). Coastal wetlands that are usually closed to tidal flushing have reduced diversity. Only 5–6 of the 19 species occur in wetlands that have long had severely impaired tidal flushing (Ballona Wetland, Deveraux Lagoon, and Malibu Creek). Those salt marshes support a near-monotype of perennial pickleweed.

6.5.5 Recommendations for Long-Term Monitoring

The shifts in data collection methods, while disruptive to data analysis, have been responsive to changes in both environmental conditions (from more to less variable) and management needs. Our early questions concerned the interannual dynamics of nesting habitat for the clapper rail. More recent concerns are issues of wetland restoration and the effects of sewage. It is probably a mistake to continue a rigid sampling program once the shortcomings of the programs are recognized. There is no single correct way to sample, even if the goal of monitoring remains constant. The variability of the system determines the number and size of the sampling units, and interannual variability cannot be known until several years of data have been accumulated.

Monitoring programs are now being required in many restoration and mitigation projects. The items to be sampled and the frequency and methods of sampling are now being set before any reconnaissance data are gathered, because regulatory agencies do not have the luxury of working out appropriate sampling schemes before writing permit conditions. An overall goal of monitoring programs for mitigation sites should be understanding how a wetland is changing through time. Not knowing how or where changes will occur makes it difficult to preset the sampling program. However, measuring interannual variability probably should take precedence over detailed measures of spatial variability, unless funding allows both to be sampled thoroughly.

The Tijuana Estuary monitoring program indicates that requirements for monitoring mitigation sites should have a broad mandate. First, the sampling programs should be undertaken as research programs in themselves. A variety of attributes should be measured until it is clear which attributes are the better indicators of system response. Different sampling units (e.g., quadrat sizes and shapes) should be compared initially to select those that show low variability within communities. Second, initial sampling should include much larger areas with far more stations than may be needed. Sampling adequacy should be reviewed every 3–5 years. Monitoring stations can be cut back once the data suggest that cutting back is permissible. Third, a hierarchical approach is indicated. A larger number of attributes and stations can be sampled with lower frequency (perhaps 5-year intervals), whereas more intensive sampling should be done annually at fewer stations.

To conclude, monitoring programs should be adaptive, responding to new information as it is gathered and to new management needs as they develop.

CHAPTER 7

Adaptive Management of Restored and Constructed Wetlands

7.1 THE NEED FOR ADAPTIVE MANAGEMENT

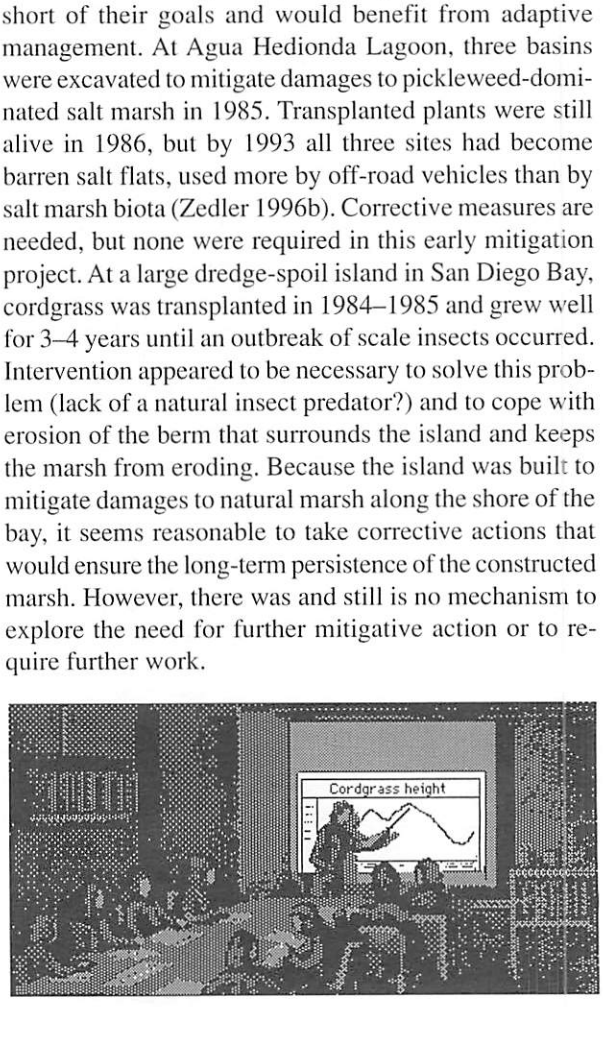
Adaptive management is the iterative approach to managing ecosystems when the methods of achieving desired objectives are unknown or uncertain to work (Holling 1978, Walters 1986). Adaptive approaches seem to be more appropriate for restoration projects than the traditional approach, which would call for implementing an entire plan without opportunity for revisions along the way.

7.1.1 Rationale for Adaptive Management

We expect restoration projects to have a greater chance of achieving their stated goals if suggestions for improved planning, implementation, and assessment are heeded. However, the process will still be experimental. Through time, developing ecosystems will experience unexpected events, new problems, or uncontrollable disturbances. Along the way, new information may become available, management actions may need to be reevaluated, and midcourse corrections may become necessary. Adaptive management can accommodate such needs. In the Famosa Slough enhancement program, algal blooms are an occasional problem, but the causal nutrient input has three potential sources and each has a unique solution. Managers need to prioritize the corrective measures after studies are done to detect the primary nutrient source (section 7.2.3). It is also possible that a restoration site will accidently attract a desired species; plans should be changed if an endangered species unexpectedly colonizes the site. The surprise appearance of the California least tern, an endangered species, necessitated reevaluation of plans to accommodate another endangered bird, the light-footed clapper rail, at the Chula Vista Wildlife Reserve. Where unknowns predominate, the approach to long-term management of constructed or restored wetlands should be an adaptive or flexible one.

Many projects go awry and require corrective measures; in fact, mitigation projects rarely develop as promised (Kusler and Kentula 1989, National Research Council 1992, Thayer 1992). Almost invariably, some component of the ecosystem or some critical function is lacking. For example, attempts to recreate tidal marshes have not led to the development of soil organic matter and nutrient levels comparable to those of natural marshes (Lindau and Hosner 1981, Craft et al. 1988, Langis et al. 1991). In addition, invertebrates in the soil have not developed comparable communities (Moy and Levin 1991), shrimp species do not use transplanted marshes as readily as natural marshes (Minello and Zimmerman 1992), and endangered birds have not been attracted to nest in transplanted cordgrass marshes (Zedler 1993).

In the San Diego area, several projects have fallen short of their goals and would benefit from adaptive management. At Agua Hedionda Lagoon, three basins were excavated to mitigate damages to pickleweed-dominated salt marsh in 1985. Transplanted plants were still alive in 1986, but by 1993 all three sites had become barren salt flats, used more by off-road vehicles than by salt marsh biota (Zedler 1996b). Corrective measures are needed, but none were required in this early mitigation project. At a large dredge-spoil island in San Diego Bay, cordgrass was transplanted in 1984–1985 and grew well for 3–4 years until an outbreak of scale insects occurred. Intervention appeared to be necessary to solve this problem (lack of a natural insect predator?) and to cope with erosion of the berm that surrounds the island and keeps the marsh from eroding. Because the island was built to mitigate damages to natural marsh along the shore of the bay, it seems reasonable to take corrective actions that would ensure the long-term persistence of the constructed marsh. However, there was and still is no mechanism to explore the need for further mitigative action or to require further work.



An adaptive approach is being used to achieve mitigation requirements at Sweetwater Marsh in San Diego Bay (section 7.2.1); the success of this interactive management program suggests that similar approaches will benefit restoration of 200 hectares at Tijuana Estuary (Entrix et al. 1991), enhancement of Famosa Slough (City of San Diego 1991), and restoration of 60.7 hectares of wetland at San Dieguito Lagoon (California Coastal Commission 1991).

7.1.2 Constraints

Understandably, agencies charged with restoring habitats will find it difficult to shift from traditional to adaptive management approaches, because of uncertainties in the costs of projects and in the time required to complete projects. In addition, a close interaction with technical experts who undertake the necessary monitoring and research, periodic meetings, and a willingness to make difficult decisions will be needed. This is unfamiliar terrain for contracting officers, and there may be resistance to the concept. Furthermore, there are no guidebooks for writing adaptive management plans for projects such as wetland restoration, so it may be difficult to find people who can prepare the necessary documents.

7.2 EXAMPLES OF ADAPTIVE MANAGEMENT

The California Coastal Commission's (1991) requirements for successful mitigation indicate how to set up criteria for successful restoration (Table 7.1). However, as plans are written, the part on adaptive management can be a major stumbling block. Problems require management to select among various options. To show how adaptive management can assist with an ongoing wetland mitigation project, we describe the San Diego Bay project and examples of adaptive management in other major restoration projects of the region.

7.2.1 Sweetwater Marsh National Wildlife Refuge: Implementing Adaptive Management

Bruce Nyden and Joy Zedler

Caltrans and the U.S. Army Corps of Engineers are being required to mitigate loss of habitat caused by construction of a new freeway interchange, the widening of an existing freeway, and excavation of a new flood control channel. The original mitigation requirements stipulated that 76 hectares be set aside. By 1986, the designated lands had still not been deeded to a public agency, so the Sierra Club and the League for Coastal Protection sued the agencies involved and won. The U.S. Fish and Wildlife Service Biological Opinion was revised, with additional mitigation requirements, a 128-hectare refuge, more rigorous standards for success, and long-term monitoring (from 1989 through compliance with

Table 7.1. Minimum Standards and Objectives for Successful Mitigation at San Dieguito Lagoon*

Minimum standards

1. Location within Southern California Bight.
2. Potential for restoration as tidal wetland, with extensive intertidal and subtidal areas.
3. Creates or substantially restores a minimum of 60.7 ha of wetlands, excluding buffer zone and upland transition area.
4. Provides a buffer zone of an average of at least 91 m wide, and not less than at least 31 m wide, as measured from the upland edge of the transition area.
5. Any existing contamination problems at the site would be controlled or remediated and would not hinder restoration.
6. Site preservation is guaranteed in perpetuity (through appropriate public agency or non-profit ownership, or other means approved by the Executive Director), to protect against future degradation or incompatible use.
7. Feasible methods are available to protect the long-term wetland values on the site, in perpetuity.

Objectives

1. Provides maximum overall ecosystem benefits, for example, maximum upland buffer, enhancement of downstream fish values, regionally scarce habitat, and potential for local ecosystem diversity.
2. Provides substantial fish habitat compatible with other wetland values at the site.
3. Provides maximum upland transition areas (in addition to buffer zones).
4. Restoration involves minimum adverse effects on existing functioning wetlands and other sensitive habitats.
5. Site selection and specific restoration plan reflect a consideration of site-specific and regional wetland restoration goals.
6. Restoration design is that most likely to produce and support wetland-dependent resources.
7. Provides habitat for rare or endangered species.
8. Provides for restoration of reproductively isolated populations of a native California species.
9. Results in an increase in the aggregate acreage of wetland in the Southern California Bight.
10. Requires minimum maintenance.
11. Restoration project can be accomplished in a timely fashion.
12. Site is in proximity to the San Onofre Nuclear Generating Station.

**Source*: California Coastal Commission 1991 the permit) to evaluate the functionality of the mitigation sites. Two marsh excavation projects were required to mitigate damages under Section 7 of the Endangered Species Act (U.S. Fish and Wildlife Service 1988).

An adaptive management approach was not required; rather it evolved from four factors: (1) mitigation criteria that required persistence of biological attributes, (2) monitoring and remedial measures to correct any problems, (3) agency biologists with foresight and willingness to work with technical experts, and (4) a research group that was interested in understanding differences between the natural and constructed wetlands (PERL). The revised standards (cf. Table 2.8) for the constructed habitat were based on perceived needs of three endangered species that were jeopardized by the projects (U.S. Fish and Wildlife Service 1988): for the lower intertidal marsh, the light-footed clapper rail; for the middle and high marsh, salt marsh bird's beak. The channels were to provide foraging habitat for the California least tern.

In 1988, PERL was asked to monitor several attributes suggested by the revised Biological Opinion, including quarterly sampling of invertebrates and fish (seining) to determine species diversity and abundance; monthly sampling of water for salinity, dissolved oxygen, and nutrients; annual sampling of vegetation for composition, percentage of cover, and cordgrass height; and monthly monitoring of soil salinity. Annual meetings were required between PERL, Caltrans, the U.S. Fish and Wildlife Service, and the U.S. Army Corps of Engineers, after dissemination of PERL's annual report of monitoring results. This was the beginning of an adaptive management program that continues to date (Fig. 7.1).

Of the mitigation requirements, only the criteria for diversity and abundance of fish were satisfied in the first three years (1989–1992). At the 1992 annual meeting, it was agreed that fish sampling could shift from quarterly to annual. Costs of fish monitoring were thus reduced, and the savings were available for assessing the population of salt marsh bird's beak in its first year of compliance monitoring. Creating naturally functioning salt marshes has proved more difficult, and an iterative approach has developed, with field comparisons of natural and constructed marshes, field experimentation to detect problems, implementation of recommended actions, and then further comparisons and experimentation.

While monitoring was ongoing, PERL obtained funding from California Sea Grant to make detailed comparisons of the constructed and natural marshes (Swift 1988, Cantilli 1989, Rutherford 1989, Zalejko 1989, Langis et al. 1991). Later research projects (funded by California Sea Grant and NOAA's Coastal Ocean Program) focused on methods of growing cordgrass to support the endangered light-footed clapper rail. The attempt to create suitable habitat for clapper rails illustrates the adaptive management approach. The density of cordgrass was eventually comparable to that found in natural marsh areas, but the plants were too short for a bird that builds a floating nest that needs to rise and fall with the tide (Zedler 1993). Soil analyses showed that the coarse soils were deficient in organic matter and nitrogen and suggested that nitrogen levels could be enhanced through soil amendments. (cf. section 5.1).

Caltrans allowed PERL to do a pilot study in a second mitigation marsh to compare different fertilization techniques. Caltrans further agreed to adopt the recommendations for soil preparation before transplanting cordgrass the year after the study. In 1990, experimental plantings showed that additions of nitrogen increased stem height most when both organic (alfalfa) and inorganic (ammonium sulfate) nitrogen were added. Hence, in 1991, Caltrans rototilled alfalfa into the cordgrass transplantation sites and added fertilizer pellets to each planting hole. PERL continued to follow the experimental plots, and in 1992, when they were 2 years old, it was clear that one-time addition of nitrogen was insufficient to induce growth of cordgrass tall enough to support nesting by clapper rails (Gibson et al. 1994). Two simultaneous experiments showed high rates of decomposition of alfalfa and rapid loss of nitrogen from sandy soils, explaining why one-time fertilization was inadequate. A subsequent experiment was designed to test the effectiveness of repeated fertilization, with variations in the timing and duration of applications. And, because the results of the initial experiment suggested that nitrogen fertilization might trigger infestations of scale insects on cordgrass, the effects of repeated applications of fertilizer on scale insects were assessed along with cordgrass responses (Boyer and Zedler 1996). Results (Boyer and Zedler, in preparation) indicated that cordgrass requires repeated nitrogen fertilization to grow tall, and Caltrans has authorized more widespread and longer term fertilization of the constructed marshes. It is hoped that the system can ultimately become self-sustaining, but it is not certain how long corrective measures will be needed.

Although the mitigation project at San Diego Bay became an adaptive management program, it might have taken a different route if several factors had been different. The major impetus was a civil suit that forced Caltrans and the U.S. Army Corps of Engineers to comply with the Endangered Species Act, specifically the Biological Opinion of the U.S. Fish and Wildlife Service (1988), which required that habitats be resilient for at least 3 years and that remedial action be taken in the year following detection of the problem. Other factors contributed to the evolution of an adaptive approach. The most important were that (1) Caltrans had a vested interest in the success of the project, (2) Caltrans biologists were willing and able to facilitate research to solve problems, (3) Caltrans designated knowledgeable wetland biologists to oversee the project, and (4) the monitoring was contracted to a laboratory that was part of an academic institution. The last factor was probably critical,

OBJECTIVE: Provide tall cordgrass for nesting habitat for clapper rails
PROBLEM: Coarse soils do not supply adequate nitrogen for cordgrass to grow tall

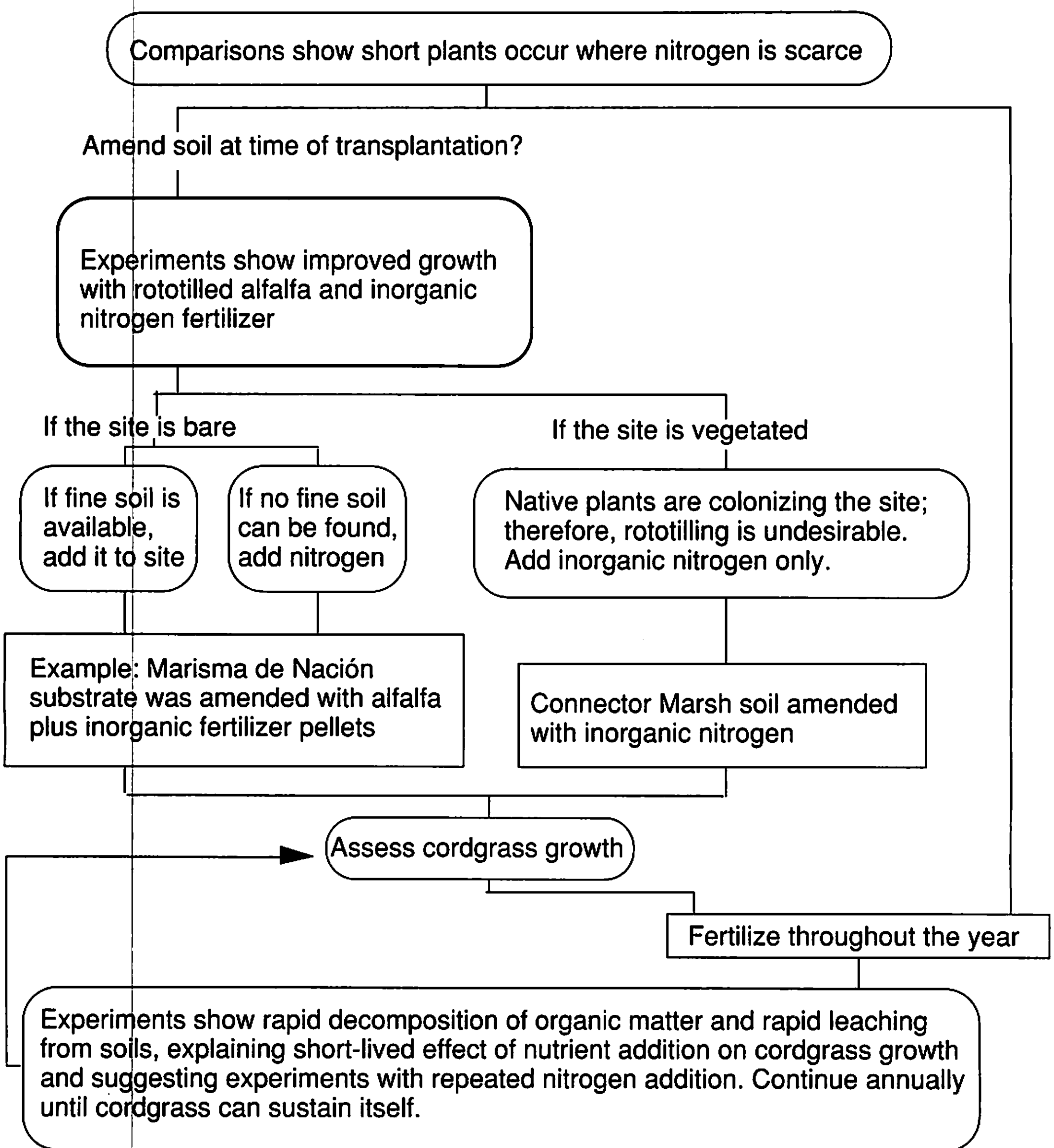


Figure 7.1 Part of the adaptive management program at Sweetwater Marsh, San Diego Bay, California, where the transplanted cordgrass is too short for nesting by clapper rails. An iterative program of research, recommendations, implementation, and further study has been evolving since 1988. Square boxes = management actions; round boxes = data obtained and recommendations made; unboxed phrases = management questions.

because the principal investigators involved were interested in understanding the problems with the constructed marshes, were able to find students to pursue different studies, and were able to secure funding for that research. A further essential ingredient may have been the ability of both Caltrans and university biologists to understand each others' views and responsibilities.

7.2.2 San Dieguito Lagoon: Establishment of an Adaptive Management Restoration Plan

The California Coastal Commission (CCC 1991) is requiring Southern California Edison to restore a minimum of 60.7 hectares of tidal wetlands at San Dieguito Lagoon as partial mitigation for the effects of continued operation of San Onofre Nuclear Generating Station. The Commission established success criteria before selection of the mitigation site, so the requirements are relatively general. Seven minimum standards were given for the site and the preliminary plan (including a guarantee that the site would be preserved in perpetuity "to protect against future degradation or incompatible land use"). In addition, the restoration plan had 12 objectives, including provision for maximum overall ecosystem benefits, a maximum upland buffer, enhancement of downstream fish values, regionally scarce habitat, substantial fish habitat, minimum adverse effects on existing wetlands and sensitive habitats, and habitat for rare or endangered species (cf. Table 7.1).

An adaptive management approach is required. Specifically, the Commission requires long-term management (through the life of the operation of San Onofre Units 2 and 3), monitoring before and after the project begins, and remediation of any failure to meet these goals and standards during the full operational years of Units 2 and 3.

The aspects of the plan that are truly innovative are the requirements for "maximum overall ecosystem benefits" rather than single-species targets, the long-term commitment of the mitigator, reliance on a scientific advisory panel to oversee the mitigation work and monitoring, and the ability to require remedial actions. The executive director of the Commission has responsibility for prescribing remedial measures. Both physical and biological performance standards are given; failure to achieve any of these standards would set in motion a series of changes in the requirements for the project and its monitoring program.

The San Dieguito Lagoon is already highly modified, with a freeway across the middle, a highway across the mouth, a racetrack, and fairground near the mouth, and urban development on both sides. An unusually large part of the river valley is undeveloped, and a planned regional park should ensure open space upstream.

One basic problem is that tidal flushing will not be easy to sustain. Like many of the region's coastal bodies of water, San Dieguito Lagoon tends to close to tidal flushing as sand builds up at the inlet over summer (cf. section 2.2). If winter rains are heavy and streamflow is sufficient, the sand is washed out and tidal flushing is restored. More often, the sand is bulldozed to improve water circulation or reduce flood potential. Regional hydrologists recommend increasing the tidal prism as a means of providing continuous tidal influence.

The inlet of San Dieguito Lagoon is far from the wetland area that will become the restored tidal wetland. The hydrologic connection with the ocean has been described as a "straw," because it is long and narrow and constrained by adjacent development. Thus, any dredging to improve the tidal prism must occur well inland of the mouth. The technology of designing inlets and tidal prisms so they can provide self-sustaining tidal hydrologic conditions is not sufficient to guarantee success. Thus, a series of trials and remedial actions likely will be needed. With increased dredging or modification of the inlet come alterations to planned habitats and effects on biota. Decisions to increase the area dredged will thus require expert consultation and careful consideration. The scientific advisory committee established by the Commission will be an essential part in the process of adaptive management.

A second problem will be providing habitat for endangered species. To date, such efforts have not been successful, at least not in the long term. One plant, salt marsh bird's beak, has been reintroduced to Sweetwater Marsh, but mainly to remnants of natural habitat, where pollination may be limiting the seed crop (cf. section 2.4). The ability of bird's beak to persist in constructed wetlands is less certain (PERL 1992).

7.2.3 Famosa Slough: Enhancement Through Adaptive Management

The California State Coastal Conservancy provided funds for the City of San Diego to develop an enhancement plan for Famosa Slough, a semi-tidal wetland. (It receives tidal flows only through a culvert). This highly modified wetland has two segments, which are separated by a four-lane city street. The two segments are further separated from the San Diego River and its ocean inlet by an eight-lane freeway. The City paid over $3 million for the innermost 8-ha segment following a long battle between the owner/developer and local citizens (the Friends of Famosa Slough). The Friends envisioned enhancements that would improve tidal flushing while maintaining the Slough's open water (lagoon) habitat for birds. With $100,000 from the State Coastal Conservancy, the City hired consultants to develop a traditional enhancement plan (PSBS et al. 1993), which provides a long list of potential actions, ranging from straightforward signage to construction of a complicated one-way tidal water flow pattern that would force water into the lagoon via a one-way flap valve, along a new channel to be dredged at the eastern and southern ends of the triangular lagoon, and

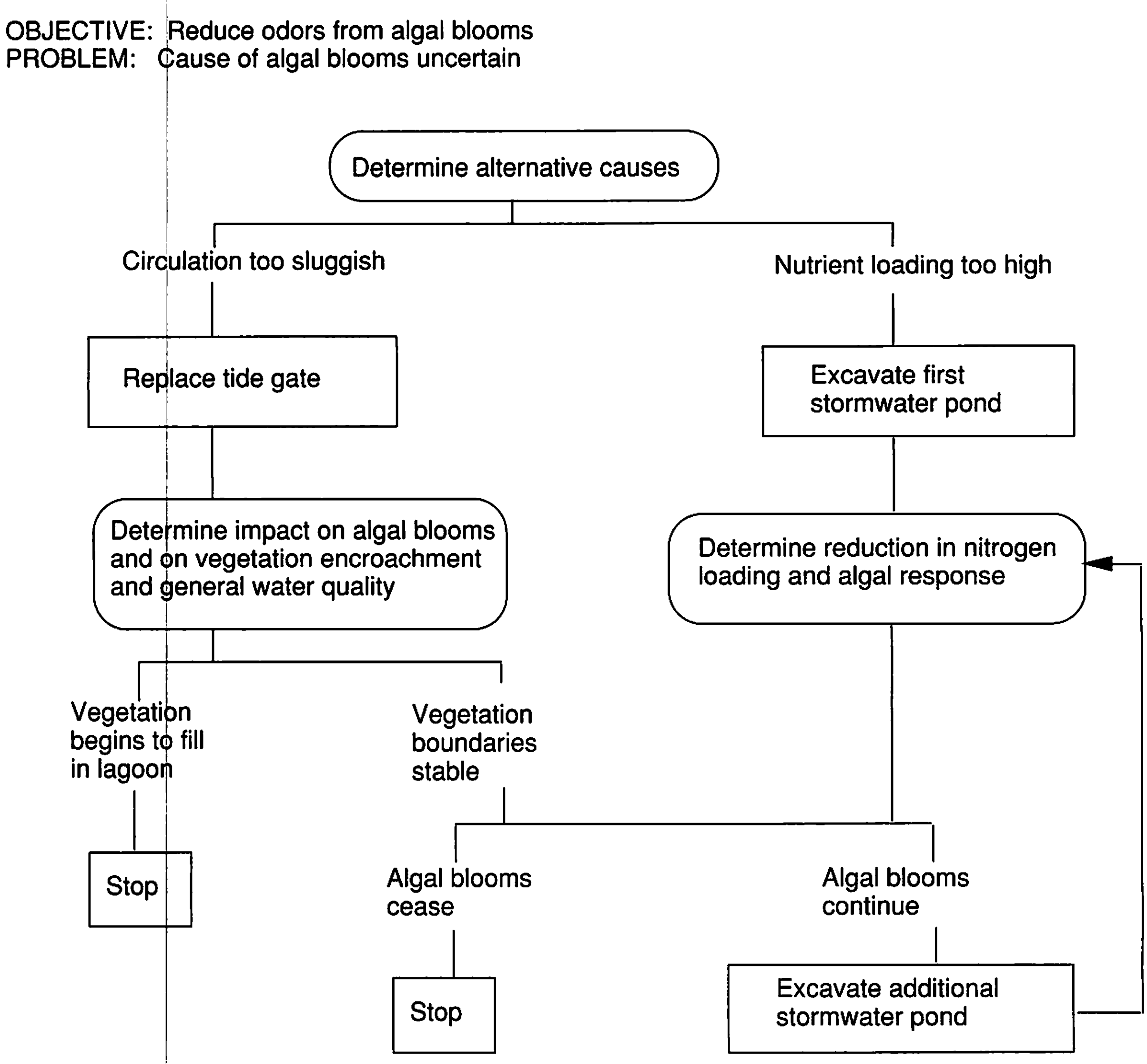


Figure 7.2 Part of the adaptive management plan being developed for Famosa Slough, San Diego, California. One of the problems to be solved is nuisance algal blooms. Because it is unclear what causes the algal blooms or what measures will reduce algal growth (circulation or loadings), an iterative approach is being recommended. Square boxes = actions; round boxes = data needs; unboxed phrases = ecosystem attributes.

out through another one-way valve.

The Friends pressed for an adaptive management approach throughout the planning process. Their strong political support for the Slough led to their becoming a part of the approval process, and the enhancement plan could not go to City Council for adoption until the Friends provided text on the adaptive management approach (City of San Diego 1991). A group of concerned professionals took on the task as volunteers and developed the necessary strategy. A decision tree for several of the significant changes proposed for the Slough is included here (Figure 7.2) to illustrate the role of adaptive management. Water quality analyses are underway to assess the source of nutrients that may trigger algal blooms (PERL, unpublished data). Depending on data collection that will answer various questions, several of the more costly and significant modifications may not need to be done.

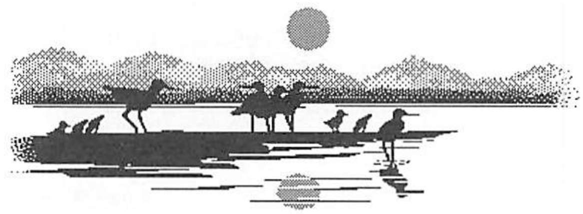
7.2.4 Other Projects in Which Adaptive Management Is Necessary or Desirable

Batiquitos Lagoon. The Port of Los Angeles selected Batiquitos Lagoon for off-site mitigation of losses to nearshore fish habitat that will occur as port facilities are expanded. Batiquitos Lagoon is closed to tidal flushing much more consistently than San Dieguito Lagoon, presumably because the former has a heavy cobble load that resists erosion during streamflow events. When the cobbles are bulldozed to lower water levels in the lagoon, closure generally recurs in a day or two. The ambitious dredging plans that have been developed to increase the tidal prism of Batiquitos Lagoon should be accomplished within an adaptive management framework. The reasons are numerous:

1. The biological functioning of the prerestoration ecosystem has not been assessed, and it is unclear what values might be irreversibly lost with excessive dredging.
2. The amount of dredging essential to providing full tidal flow is uncertain.
3. The adopted plan takes a conservative approach from the engineering perspective (the alternative with minimum dredging was rejected) and leaves the risk to biota that may be unnecessarily threatened by excessive dredging.
4. The need for maintenance dredging is anticipated.

Mission Bay. The City of San Diego (Wallace Roberts and Todd et al. 1992) updated its master plan for Mission Bay, a 1,860-hectare recreational park with about 12 hectares of remaining salt marsh. Included in the plan is the reconstruction of 28–36 hectares of salt marsh to enhance natural resources and assist in improving the quality of water entering the bay from the small but urbanized watershed. When the time comes to expand the wetlands, it should be done with an adaptive management approach for the following reasons:

1. The size and configuration that would most likely persist given the altered tidal and stream hydrology is unknown.
2. The sizes, shapes, and qualities of habitat required to support different salt marsh communities and food webs are unknown.
3. The most effective configuration for improvement in water quality is unknown.
4. The effects of future dredging to maintain boating channels are uncertain.
5. Mechanisms for protecting wetlands from boating and other recreational activities have not been developed.
6. The requirements for type and size of buffers between the marsh and adjacent urban developments are unknown



7.3 CONCLUSIONS

Adaptive management is a useful approach to restoration because it offers a mechanism for making decisions when it is uncertain what management actions will bring about the desired goals. It should aid several restoration projects that are proposed for Southern California coastal wetlands. Although the concept has been introduced in a few restoration plans, it is not yet fully developed as a management strategy. Initial attempts have suggested the use of decision trees to organize major management options, for which decisions must be based on monitoring data or experimental results.

Perhaps the most important element of a successful adaptive management program is the interaction between the technical advisors and the project manager. Without access to researchers who can develop the necessary experiments as needed and without their willingness to translate complicated information for project managers, the scientific advice will have limited utility. And without some understanding of ecosystem functioning on the part of the project manager, the interaction may not be workable. Thus, appropriate personnel on both the technical and managerial sides are critical to the success of adaptive management.

It remains to be seen whether adaptive management programs can be conducted under innovative contracts between agencies and consulting. If flexibility and ability to follow opportunities are the key ingredients for adaptive management, then agencies will need to accommodate contracts that may change in direction and magnitude from year to year.

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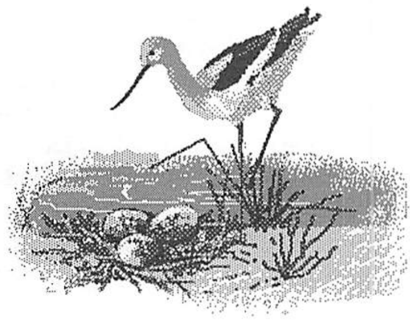

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ISBN 1-888691-02-6