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Can We Restore Our Salt Marshes? Lessons from the Salmon River, Oregon

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Introduction

Relative to many areas of the United States, the Pacific Northwest is endowed with few wetlands; however, substantial wetland resources have been lost in the region since European settlement. An estimated 3.6 million acres of all types of wetland in Washington and Oregon existed in the 1780s; of these, 35.4% have been converted to other uses or severely degraded (Dahl 1990). Historically, agricultural land development has been the main cause of wetland conversion, although in recent times losses can increasingly be attributed to industrial and commercial development (Tiner 1984). Coastal salt marshes, close to ports and population, are especially vulnerable, with losses commonly between 50 and 90% in individual estuaries (Bortleson, Chrzastowski, and Helgerson 1980; Boulé and Bierly 1987; Thom and Hallum 1990). Ironically, as wetland conversion has continued, recognition of their ecological and social values has increased. Indeed, wetlands provide many important functions, including flood control benefits, water quality enhancement values, habitat and life support for fish and other wildlife, areas for recreation, and opportunities for scientific research (Mitsch and Gosselink 1986; The Conservation Foundation 1988).

Since the early 1970s, in response to continuing loss of wetlands and the ecological services that they provide, many attempts have been undertaken to restore degraded wetlands or create new ones. Most of these attempts have been carried out to compensate for losses related to development (Kusler and Kentula 1990). Strategies for compensation were formalized through federal mitigation policy. Initiated in 1958 under the Fish and Wildlife Coordination Act, mitigation has since become increasingly codified (Blomberg 1987). Recently, the Department of the Army and the United States En-

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vironmental Protection Agency established policy and procedures through a "Memorandum of Agreement" for the kind of mitigation required for standard permits under the Clean Water Act Guidelines [33 CFR 325.5(b)(1)]. Accordingly, the Army Corps of Engineers "... first makes a determination that potential impacts have been avoided to the maximum extent practicable; remaining unavoidable impacts will then be mitigated to the extent appropriate and practicable by requiring steps to minimize impacts and, finally, compensate for aquatic resource values" (Department of the Army and Environmental Protection Agency 1990).

Compensatory mitigation under the Clean Water Act Guidelines includes *restoring* existing degraded or former wetlands or *constructing* wetlands on sites that have not supported wetlands in recent centuries. In some instances, *enhancement* of particular wetland values or functions (e.g., selectively increasing aquatic habitat for waterfowl) is an acceptable, although less desirable, compensatory measure (Kruczynski 1990). In practice, the Memorandum of Agreement establishes a sequence of steps aimed at avoiding or diminishing environmental damage. Of the three compensatory strategies, restoration is preferred because success is more likely to be achieved and the outcome of the restoration can be more easily predicted (Kruczynski 1990; Kusler and Kentula 1990; Department of the Army and Environmental Protection Agency 1990).

Another reaction to the loss of wetlands and their values is the recent attempt by the President, legislators, wetland managers, and environmentalists to formulate and implement national and state policy with the goal of "... no overall net loss of the nation's remaining wetlands base, as defined by acreage and function" (The Conservation Foundation 1988). Under this framework, restoration and creation are not mitigation measures, but they are recognized as a means of stemming wetland loss, as well as increasing our wetland resource base. However, whether for the purposes of mitigation or for carrying out the policy of "no net loss," techniques and results of restoration and creation often have been unsuccessful (Race and Christie 1982; Race 1985; Josselyn, Zedler, and Griswold 1990; Lewis 1990).

A common practice in compensatory mitigation along the West Coast is to return former diked pasture to salt marsh by removing dikes and re-connecting to estuaries. In this paper we evaluate the progress of salt marsh restoration in the Salmon River estuary along the north central Oregon coast ten years after breaching dikes that enclosed a pasture. First, we assess restoration by examining changes in a limited number of wetland characteristics including: plant species, plant communities, elevation of the site, the role of salinity and soil texture, width and depth of creek cross-sections, and estimated above-ground net primary production (rate at which available energy is incorporated in the salt marsh). From this assessment, we develop some guidelines to aid wetland managers in carrying out coastal salt marsh restoration actions. The results are applicable from Humboldt Bay in northern California, north to British Columbia. We share our experience with the aim of helping wetland managers achieve successful restoration projects in the Pacific Northwest.

Salmon River Estuary

An unusual opportunity to evaluate salt marsh restoration presented itself when, in 1974, the U.S. Congress established the Cascade Head Scenic-Research Area along the north-central Oregon coast (P.L. 93-535). The U.S. Forest Service administers this specially designated area which includes the Salmon River estuary. A long-term goal for the area is the ". . . revitalization and restoration of the Salmon River estuary and its associated wetlands to a functioning estuarine system free from the influences of man" (U.S. Forest Service 1976). Prior to restoration, about 75% of the Salmon River salt marshes had been isolated from estuarine circulation by dikes for conversion to pasturage. The specific intent of the Forest Service was to return the estuary ". . . to its condition prior to the existing diking and agricultural use" (U.S. Forest Service 1976).

In 1978, a dike enclosing a 52-acre pasture along the north shore of the Salmon River estuary was removed by the Forest Service. The objective of dike removal was to replace the pasture with salt marsh. This project was a *deliberate* restoration action. Although not targeted for mitigation, the Salmon River experience is valuable to wetland managers as they try to restore comparable coastal salt marsh areas under the mitigation requirements of the Clean Water Act and state statutes such as Oregon's Removal-Fill Law (ORS 196.800 *et seq.*) and Mitigation Bank Act of 1987 (ORS 196.600 *et seq.*).

Restoration was monitored for ten years after dike removal in 1978, allowing us to address the question of whether it is reasonable or even possible to precisely replace a highly altered ecosystem with a system approximating the pristine one that may have existed on site. Return of a wetland to "original conditions" is a common objective of restoration, but this goal is difficult, if not impossible, to meet. Zedler (1984) identified three problems in achieving this objective: (1) Usually there is little knowledge of a wetland's original character; (2) disturbances to the wetland may be irreversible; and (3) both intact and disturbed wetland ecosystems are constantly changing.

Each of these three problems showed themselves in the Salmon River. First, we lack accurate knowledge of conditions prior to agricultural use. The majority of the Salmon River marshes were grazed and harvested for hay since the 1920s. Dike construction to extend pasturage occurred principally in the early 1960s (Mitchell 1981). Based upon examination of historic air photos and observations of remnant undiked marshes, we *presume* that: (1) the diked pastures had been high salt marsh dominated by tufted hairgrass (*Deschampsia caespitosa*), Baltic rush (*Juncus balticus*), and Pacific silverweed (*Potentilla pacifica*); (2) they were sufficiently high that they were seldom flooded by salt water in summer; and (3) they were deeply dissected by tidal creeks. Existing undiked salt marshes provide what evidence there is of the original character of the diked parcels. These relatively intact wetlands serve as "controls" and act as imperfect "yard sticks" against which the restored marsh can be measured. Second, historic diking is a profound disturbance with potentially irreversible consequences to the progress of the restoration. Third, although the remnant, relatively intact control marshes show changes typical of natural salt marshes, they also have been indirectly disturbed by diking.

The Forest Service acquired about 80 acres of diked pasture and adjacent high salt marsh on the north shore of the estuary in 1978 (Fig. 1). The pasture was created in 1961 by enclosing high salt marsh behind a dike fitted with a tide gate. In early September 1978, about half of the 5,000-foot dike was removed. From 1977 to 1980, Diane Mitchell, a graduate student in botany at Oregon State University, worked in this area. Mitchell established an intensive sampling system of permanent plots and collected base-line data in the diked pasture and in the adjacent high marsh "controls." For her doctoral dissertation she analyzed the initial restoration changes from 1978–1980 (Mitchell 1981). In this paper, we combine Mitchell's earlier data with our own collected in 1988.

Vegetation change was monitored at 115 one-meter-square permanent plots along 20 transects and at 450, 0.1-meter-square temporary plots, prior to dike removal in 1978 and after breaching in 1979, 1980, 1981, 1982, 1984, and 1988. We used data from the field to define plant communities that were determined objectively by the classification program TWINSPAN, referring to Two-Way Indicator Species Analysis (Hill 1979). Restoration site elevation was systematically surveyed with respect to a tidal datum in 1978 and the survey repeated ten years later. Salinity was measured in late September 1988; soil samples were collected and analyzed for soil texture and soil organic content. Sediment accretion (accumulation) since 1978 was measured above an artificially placed sand layer.

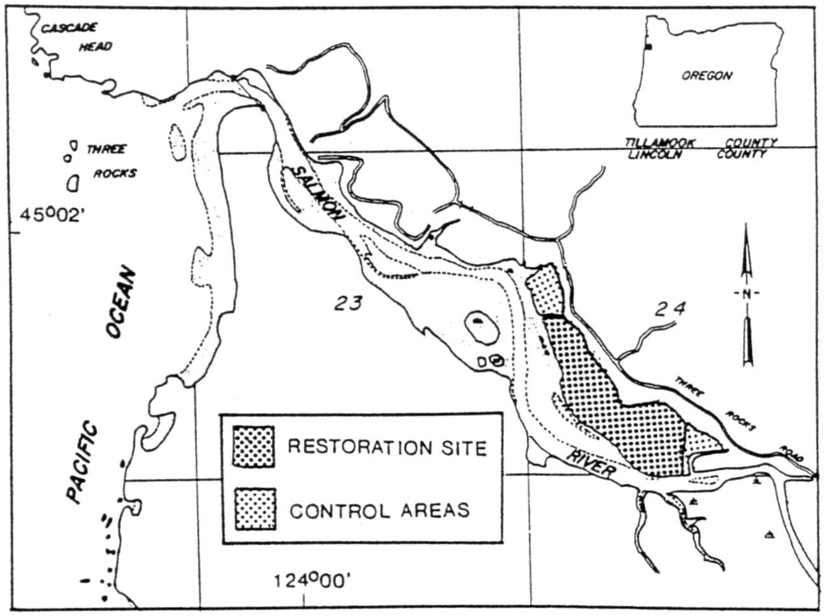

Fig. 1. The Salmon River estuary restoration site and adjacent intact control salt marshes along the north central Oregon coast.

Elevations also were measured across 47 cross-sections along eight tidal creeks to determine the extent of creek erosion over the ten-year restoration period. Further details of the methods are given in Mitchell (1981) and Frenkel and Morlan (1990).

Restoration of a Salmon River Salt Marsh

Vegetation

The relative amount of cover for four plant species groups changed radically over the restoration period from 1978 to 1988 (Fig. 2 and Table 1). *Upland pasture* species were immediately killed or heavily stressed after saltwater incursion. *Residuals,* which are marsh plants that tolerate both saline and freshwater conditions, persisted on the restoration site after dike-breaching. Dominant among these plants, are Pacific silverweed and bentgrass (*Agrostis alba*). Cover and biomass (amount of living matter) of residuals diminished since the restoration. *Ephemeral colonizers,* which are invading annual species that are present for only a few years (including brass buttons, *Cotula coronopifolia*), appeared the year after dike removal but dropped out by 1984. *Permanent colonizers* were absent in the pasture but invaded

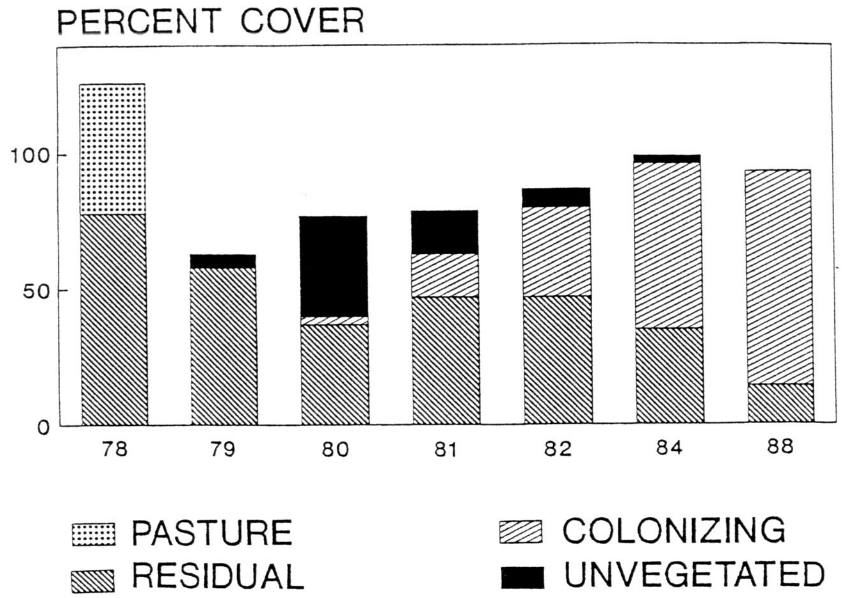

Fig. 2. Average percent canopy cover of pasture, residual and colonizing (both persistent and ephemeral) species, and percent bare ground in the Salmon River estuary restoration site, north central Oregon coast, 1978 to 1988. The dike was breached after 1978 survey.

after saltwater intrusion; they include Lyngbye's sedge (*Carex lyngbyei*), pickleweed (*Salicornia virginica*) and salt grass (*Distichlis spicata*). These species first showed up in 1981 and dominated the restored marsh by 1984. Unvegetated soil or bare ground, most extensive the second year after breaching, had essentially disappeared by the sixth year. Adjacent control plots displayed fluctuations in species cover over the same period (Table 1) but the basic species composition remained the same.

After dike breaching in 1978, upland pasture plant communities began to change and were rapidly replaced in 1979 by salt marsh communities and much bare ground. By 1984, salt marsh communities began to develop, displacing bare ground and stressed residual species. Ten years after dike removal, the marsh was dominated by four important communities with greater than 90% occurrence in restored plots (Fig. 3). Prominent plant communities included: two intertidal communities (a Lyngbye's sedge community and a pickleweed–saltgrass community) and two mid-tidal communities (a Lyngbye's sedge–Pacific silverweed community and a bentgrass–Pacific silverweed community). The two intertidal communities

TABLE 1. Mean percent cover and percent frequency of selected species and percent bare ground in the restoration area and control in the Salmon River study site, 1978 to 1988. Variation in cover and frequency from year to year in the control reflects different sample size in different years. Number before slash in body of table is mean percent cover; number after slash is frequency. Species status is shown by U = upland pasture, R = residual, and C = colonizer (ephemeral and persistent).

Area		Restoration				Control		
Year		1978	1980	1984	1988	1978	1982	1988
N		49	49	49	49	244	47	29
Species and Status								
Trifolium repens	U	7/13	—	—	—	—	—	—
Ranunculus repens	U	6/7	—	—	—	—	—	—
Poa trivialis	U	5/28	—	—	—	—	—	—
Potentilla pacifica	R	45/69	15/58	10/57	9/41	21/56	20/62	27/59
Agrostis alba	R	44/88	23/67	15/59	9/29	—	8/34	5/42
Juncus balticus	R	2/16	5/38	7/27	7/27	22/71	51/79	34/66
Hordeum brachyantherum	R	2/16	4/17	2/18	1/18	4/25	—	2/38
Carex lyngbyei	C	—	3/29	51/71	67/82	3/26	6/21	10/21
Salicornia virginica	C	—	4/40	11/33	10/18	3/9	10/19	6/14
Triglochin maritimum	C	—	1/10	2/20	1/8	2/27	5/21	2/14
Deschampsia caespitosa	C	—	1/10	1/10	1/10	5/23	5/34	15/44
Distichlis spicata	C	—	—	8/27	12/24	7/36	12/43	7/28
Spergularia marina	C	—	7/42	<1/6	—	—	—	—
Bare Ground (% cover)		0	40	4	0	0	0	0

characterized over 80% of the restored area and the mid-tidal communities were beginning to develop at the upland fringe of the restored wetland.

Physical Environmental Factors at the Salmon River Site

A number of environmental factors control hydrology, vegetation, and general ecosystem functioning of coastal wetlands. Among these factors are surface elevation, sedimentation, salinity, soil texture, and creek morphology. In this section, we summarize principal changes and status of these conditions in the Salmon River restoration site.

Surface elevation is the principal control of marsh hydrology and vegetation because the height of the land affects tidal flooding and the presence of water and, in turn, determines the type of salt marsh vegetation. In the study area, the surface elevation was about 35 cm (14 in.) lower in the restored area in 1988 than in the flanking controls (Fig. 4). The reason for a lower elevation was that the diked pasture had subsided over the diking period, from 1961 to 1978, due to buoyancy loss, compaction, and organic soil oxidation.

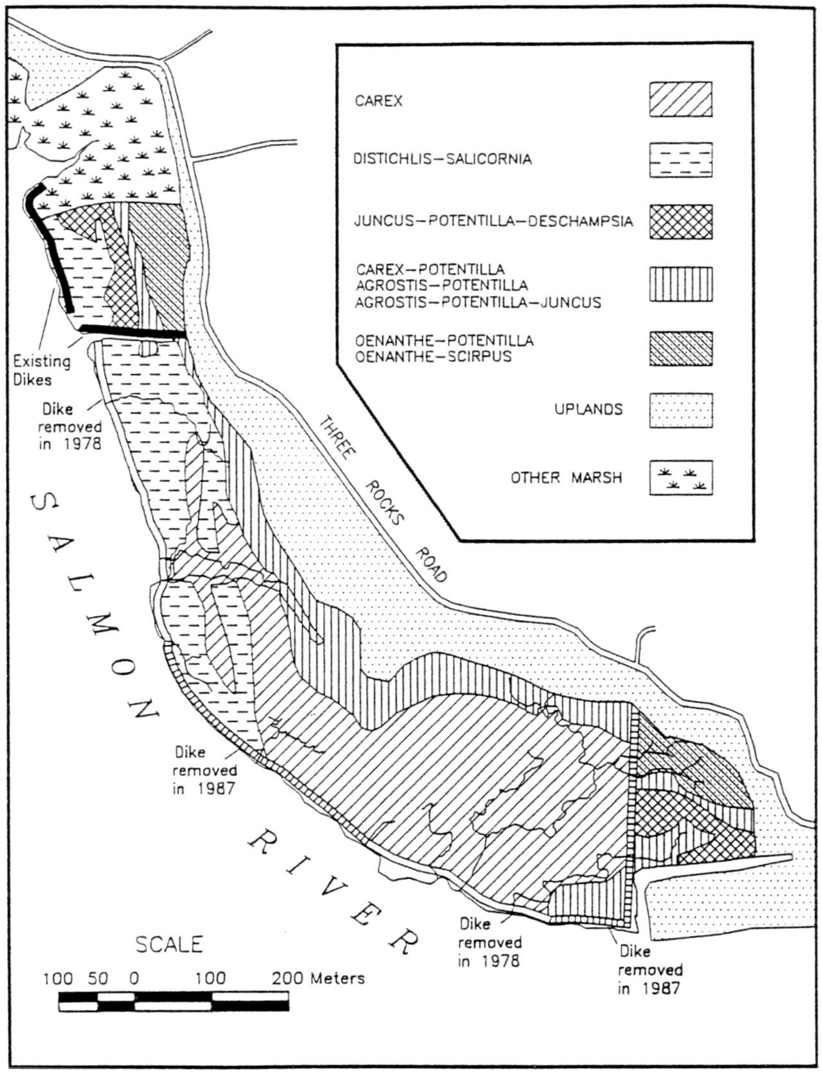

Fig. 3. Salt marsh community groups in the Salmon River estuary restoration site, north central Oregon coast, and adjacent intact controls, 1988.

Mitchell placed sand layers on the pasture surface in 1978 in order to mark the 1978 surface for locating at a future time. In 1988, our measurements of accreted sediment above the sand layer revealed that the lowest areas increased in elevation at a greater rate (5 to 7 cm per decade) than the highest areas (3 to 4 cm per decade). We found that the restored marsh surface was building up by a combination of sedimentation and soil swelling. This recovery from

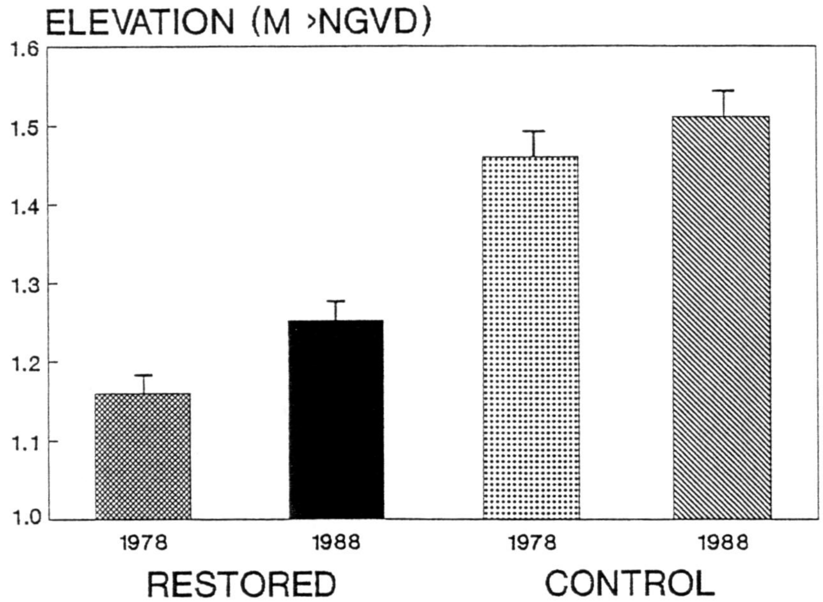

Fig. 4. Average elevation of restored Salmon River estuary site, north central Oregon coast, and adjacent control areas in 1978 (before dike breaching) and 1988 (ten years after restoration).

subsidence is expected to take five decades or more, based on the rate of recovery over the first ten years; this assumes some decline in accretion rate as the marsh surface elevates.

Salinity and soil texture control the presence of salt marsh species. Soil texture also affects soil saturation (Mitsch and Gosselink 1986). We measured soil salinity in late September when salinities are anticipated to be highest in Pacific Northwest marshes (Hutchinson 1989). As expected, salinities were higher close to the river than close to the upland. We analyzed our data using the non-parametric Spearman's rank correlation as a statistical test because our data were not normally distributed. As tested by Spearman's rank correlation (r_s), percent cover of Lyngbye's sedge ($r_s = -0.42$, $P = 0.003$) decreased with increases in salinity. However, percent cover of pickleweed ($r_s = 0.59$, $P = 0.002$) and percent cover of saltgrass ($r_s = 0.66$, $P = 0.000$) increased proportionally with salinity. Species composition was also directly related to soil texture, which we measured in the laboratory by separating the sand fraction from the silt and clay in the soil. For example, pickleweed cover ($r_s = 0.48$, $P = 0.001$) and saltgrass cover ($r_s = 0.56$, $P = 0.000$) were higher on sandy soils than silty soils. In contrast, percent cover of Lyngbye's sedge ($r_s = -0.42$, $P = 0.005$), established on siltier soils, diminished as percent

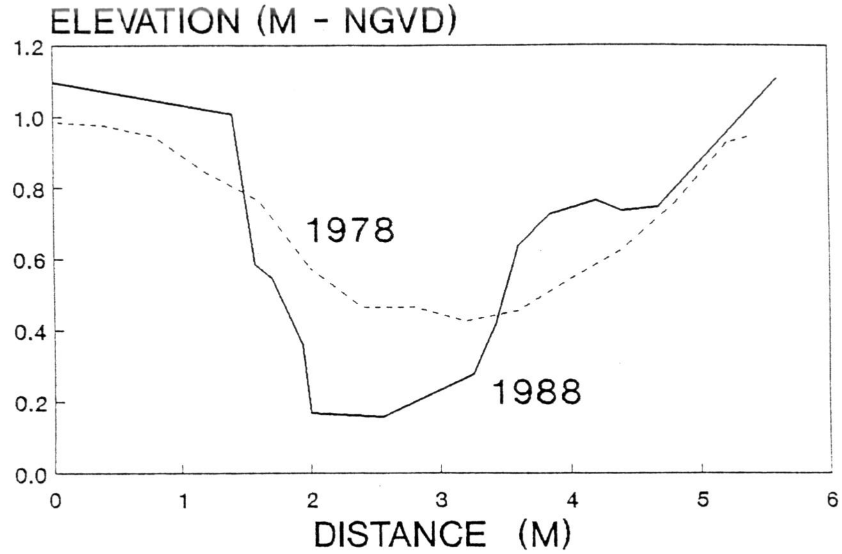

Fig. 5. Elevations across a typical creek cross-section in the Salmon River restoration site, north central Oregon coast. Erosion is shown from 1978 (before dike breaching) to 1988 (ten years after restoration). The depicted cross-section is located on a major creek system entering site from the south, about 125 m northeast of the dike cut.

sand increased. In summary, the downstream portion of the restoration site was sandier and more saline and the upstream portion was siltier and more brackish; species composition reflected this pattern with pickleweed and saltgrass occupying the downstream area, and sedge occupying the upstream area.

Tidal water principally enters a salt marsh by winding tidal creeks. After diking in 1961, the Salmon River creeks became clogged with sediments derived from cattle-trampled creek banks and because tides could no longer wash sediments from creeks. Following dike removal, tidal water began to scour filled-in creeks and these creeks began to erode under daily tidal pulses, becoming deeper and narrower (Fig. 5). At the same time that creeks became deeper, creek bank surface elevation was increasing about 6 mm per year by local sedimentation. The net effect was that creeks typically deepened 20-60 cm over the ten-year restoration period.

Salt Marsh Productivity

Wetland ecosystems provide a wide range of services including habitat and life support for fish and other animals. An important

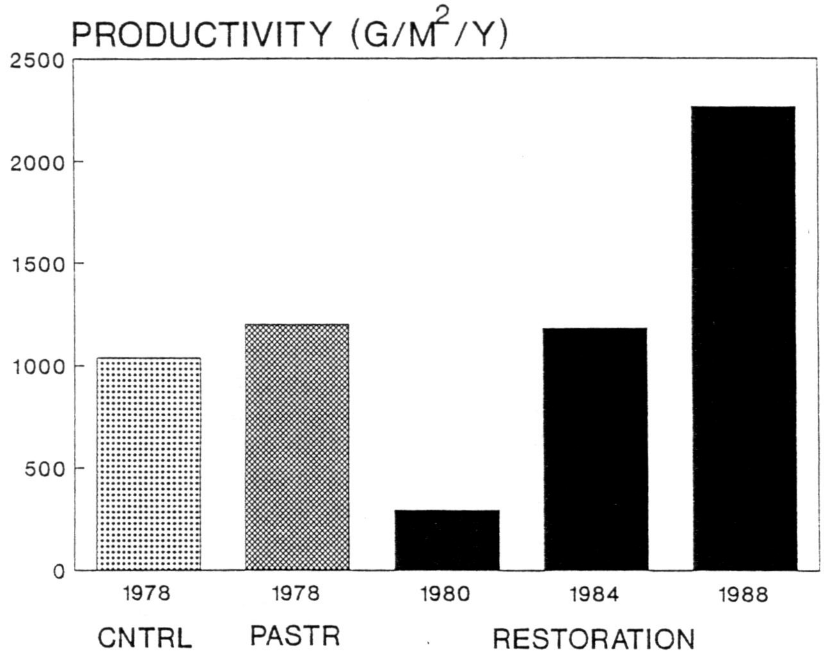

Fig. 6. Above-ground net primary productivity in the Salmon River restoration site, north central Oregon coast, and control from 1978 (pasture) to 1988. Primary productivity measures the rate at which available energy is incorporated into an ecosystem.

gauge of life support is primary productivity (a measurement of the rate at which energy is incorporated into a salt marsh). In the restored site, net primary productivity was estimated from a single harvest of above-ground vegetation following the method of Kibby, Gallagher, and Sanville (1980). Figure 6 shows trends in estimated above-ground net primary productivity in the control, pasture, and restoration area from 1978 to 1988. The pre-restoration diked pasture and unaltered control salt marshes exhibited about the same productivity of 1,000 to 1,200 g/m^2/yr dry weight, comparable to, or slightly higher than, other Northwest high salt marshes (Hutchinson 1986). Ten years after dike removal, primary productivity in the restored low marsh was 2,300 g/m^2/yr, almost double that of the pasture and control. Such a high productivity suggests an enhanced life support to the estuary since salt marsh primary productivity forms the energy and food-base for consumers such as fish and bottom-feeders. We attribute the elevated productivity to the response of the marsh system to imported nutrients derived from fresh sediments, nutrient releases from decomposing organic matter, and preemption of the site by a few adapted plant species such as Lyngbye's sedge, salt grass, and pickleweed. We anticipated higher net primary productivity for this recently restored site because this is typical of a developing ecosystem under conditions of disturbance, high nutrient availability, and low metabolic energy requirements (Odum 1969).

Lessons and Conclusions

The Forest Service goal was to restore diked pasture land in the Salmon River estuary to a naturally functioning salt marsh ecosystem. This is often the same goal when restoration is the selected strategy in compensatory mitigation. The goal was met in the limited sense that the restored salt marsh now consists of typical Pacific Northwest salt marsh communities; tidal exchanges are complete; creeks now provide habitat for juvenile fish; and the marsh is highly productive. The more rigorous restoration intent—of returning diked pasture to its *original* high salt marsh condition—was not achieved. Although the restoration project in the Salmon River fell short of a rigorous restoration goal, a number of important lessons have been learned. The following six points summarize what we discovered from this project.

1. Hydrological Connections Should Be Reinstated

Full tidal connection with the estuary by dike removal and creek excavation is probably the most important step to successful salt marsh restoration. The importance of hydrological connection has been repeatedly emphasized by other researchers (Zedler 1984; Kusler and Kentula 1990). Estuarine water mostly enters a high marsh via sinuous tidal creeks, which must be wholly reconnected with the estuary. During diking, relic creek systems may have become filled with sediments or otherwise degraded, in which case, creek excavation may be required to restore tidal connection.

Besides entering a salt marsh by creeks, tidal water may directly penetrate low salt marshes from the open estuary by gradually spreading over the wetland. However, where high marsh abuts the bay or river, as is the case in the Salmon River, a natural levee separates marsh from bay. It is this natural levee that usually forms the base upon which a dike is built. We explored whether the dike should be completely removed, or if it would be sufficient to just connect the creeks with the estuary. In 1978, about half of the Salmon River study site outer dike was removed. In 1988, when tides exceeded Mean Higher High Water, we observed flooding over the residual natural levee; at lower tides, flooding was by creeks. With monthly and seasonal over-levee flooding and associated sedimentation, restoration will be more rapid if dikes are completely removed than if only creeks were opened. We conclude that for full, rapid hydrological restoration, complete dike removal is preferable to partial dike removal, and removal of a few dike sections is better than merely opening creek connections; however, partial restoration may be achieved by just making creek connections.

2. Pre-Restoration Surface Elevation Must Be Determined

To accurately predict the kind of wetland to be restored, two things are needed: (1) a precise elevation survey tied to a primary bench mark (mean sea level), and (2) knowledge of the relation of the bench mark to a local tidal datum. In the Salmon River, over 17 years, the pasture surface subsided 35 cm (14 in.). After tidal reconnection, low rather than high salt marsh plant communities developed on the locally subsided surface. We found that the kind of species established could be predicted by knowing elevation and salinity. For example, on recently levelled dikes that were just 20 cm (8 in.) above the subsided pasture surface, a tufted hairgrass high salt marsh community was beginning to develop, as we anticipated, based on surface elevation.

3. Salinities and Soil Texture Should Be Known

The species established on the Salmon River restoration site were related to soil salinity and texture. When these characteristics are known beforehand, it is easier to predict the species composition and ultimate direction of restoration. This observation is supported by Hutchinson (1989), who surveyed salinity tolerance of Pacific Northwest salt marsh species, and by Dawe and McIntosh (1991), who observed that the level of salinity was a major factor affecting species establishment for a marsh restoration project in the Englishman River estuary, British Columbia.

4. Planting May Be Unnecessary

No transplanting was done in the Salmon River; yet, a full array of native salt marsh species became established naturally. In this case, naturally occurring and dispersed local propagules were readily available, and hydrological conditions were adequate for natural salt marsh development. Under some circumstances, local propagules may be in limited supply. In the absence of local seed sources or when hydrological conditions are marginal for natural establishment, it may be necessary to plant shoots or cuttings of desired species (Josselyn and Buchholz 1984; Zedler 1984). Our results in the Salmon River mirrored Dawe and McIntosh's (1991) experience in coastal British Columbia where marsh restoration depended exclusively on the natural influx of local propagules.

5. Monitor at Least for Ten Years

Repeatedly, researchers have stressed the importance of carefully planned and conducted quantitative monitoring (Race and Christie 1982; Josselyn and Buchholz 1984; Zedler 1984; Race 1985; Kusler and Kentula 1990). For example, Zedler (1984) recommended "... detailed monitoring of initial marsh development be followed by long-term analysis of ecosystem structure and functioning." Erwin (1990) suggested an intense program for evaluating wetland restoration and recommended annual monitoring for five years.

It was not surprising to find many transformations in the Salmon River wetland as it recovered over the ten-year restoration period. We found rapid changes occurring in species cover, community structure, hydrology, and function over the first five years after dike removal. Rates of change diminished thereafter, but ten years after dike breaching, we still observed change in species cover, biomass, and surface elevation. Wetland characteristics continue to change after ten years, although at a slower rate than initially. In some cases, radical changes have been reported after considerable time; for example, Dawe and McIntosh (1991) found that more saline tolerant species began to replace more brackish species seven years after dike breaching in British Columbia's Englishman River estuary. We recommend annual or biennial monitoring in the first four years of a restoration action, and subsequent monitoring at three-year intervals for 10 to 15 years. Monitoring at these frequencies should help determine if corrective measures are necessary, such as altering creek connections, manipulating surface elevation, and planting different species.

6. Establish Explicit, Realistic Goals

In the Salmon River site, replacement of upland diked pasture by pristine high salt marsh is probably an unrealistic goal in the short run of several decades. A more *realistic* short-term goal is to reestablish a functioning salt marsh ecosystem consisting of natural plant communities. These communities should develop in close relationship to daily pulses of tidal water and provide life support for fish and other estuarine-related animals, as well as contribute other ecological services commonly provided by intact wetlands. This is what is present in the restoration site today.

It is impossible to predict whether or not the restored area will ultimately return to high marsh with its characteristic plant communities. Barring catastrophic relative sea-level changes (Darienzo and Peterson 1990), we believe that many high marsh characteristics will develop because we continue to see changes in the low marsh plant community composition, and the site's upland margin is increasingly becoming dominated by a number of high marsh species (Fig. 3). These species changes are coming about because sediments are being deposited at a greater rate at low elevations in the restored wetland than at high elevations, and surface elevations are increasing at a greater rate in the restoration area than in the controls.

In 1991, 13 years after dike removal, visual assessment and observations indicate that highly productive low salt marsh communities prevail where formerly less productive high salt marsh dominated. Tidal water circulates with associated interchanges of organic and inorganic materials. In the long term, however, prospects for complete high salt marsh restoration are problematic. The restored area is more than a foot lower than surrounding high marsh, so it will take at least 50 years for the surface to be high enough to support original marsh vegetation. In the meantime, surrounding natural marshes will themselves change. Future marsh and estuarine character will necessarily reflect a heritage of altered hydrology.

Acknowledgments

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