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RESTORATION OF THE SALMON RIVER SALT MARSHES: RETROSPECT AND PROSPECT

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Final Report to the U.S. Environmental Protection Agency
Funds supporting this study have been provided by Region 10
of the U.S. Environmental Protection Agency, Seattle, WA

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PREFACE

In 1978 the U.S. Forest Service breached a dike on the north shore of the Salmon River estuary to reestablish a natural salt marsh in a diked pasture. Diane L. Mitchell, a graduate student at Oregon State University, initiated a detailed study of the restoration of the salt marsh ecosystem in 1977. Her work was completed in 1981. In this report, we summarize the status of the restoration in 1988, eleven years after dike removal, and discuss prospects for total restoration to conditions prevailing prior to human alterations.

The U.S. Environmental Protection Agency, Region 10, funded the present study under Assistance Grant No. X-000366-01-0. The Agency's objectives were to more fully document the Salmon River salt marsh restoration, more than a decade after initiation, and to develop recommendations for future coastal salt marsh restoration projects in the Pacific Northwest.

We have been fortunate in the continued support and cooperation of project officer, William M. Riley, Chief of the EPA Region 10 Wetlands Section of the Wetlands Division.

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SUMMARY

We assessed restoration of a 21 ha diked pasture in the Salmon River estuary to a naturally functioning estuarine salt marsh in 1988, eleven years after partial dike removal in 1978. Diane Mitchell (1981) collected base line data, established an intensive sampling system of permanent plots in the diked pasture and flanking "intact" control marshes, and analyzed restoration progress from 1978 to 1980. Our report continues Mitchell's earlier research by evaluating the composition, structure, function, and long term prospects for the restored wetland.

Composition

Species composition changed over the restoration period. Pasture species were mostly killed upon intrusion of salt water. Potentilla pacifica and Agrostis alba, prominent in the pasture, remained as residuals and continued as important species in the developing marsh. Unvegetated soil was most extensive (ca. 40% cover) in the second year after dike removal and by the sixth year (1984) was again negligible. Ephemeral colonizers (Cotula coronopifolia and Spergularia marina) established in the first and second year after breaching, but by 1984, were negligible. Carex lyngbyei, Salicornia virginica, Distichlis spicata, and Juncus balticus were important colonizers that, by 1988, characterize many of the developing salt marsh communities.

Pasture communities were replaced in the restoration area by five salt marsh communities. A low elevation, nearly monotypic C. lyngbyei community occupied much of the upriver two-thirds of

the site. The remaining downriver one-third, was largely dominated by a low elevation S. virginica-D. spicata community. The upland margin of the site was marked by communities with C. lyngbyei, A. alba, P. pacifica and J. balticus. There was little resemblance between the natural low elevation salt marsh communities and those high elevation communities characterizing the two flanking control sites.

Structure

Surface elevation, a primary control on marsh hydrology, composition and function, was 35 cm lower in 1988 in the restored area than in the flanking controls. This lower elevation was a consequence of 17 years of diking and associated subsidence. The lowest areas of the restored wetland were upriver in the area of Carex dominance. Elevation profiles across the restored wetland showed that the lowest elevations were increasing in elevation due to sedimentation and buoyancy at a greater rate (8 to 10 cm per decade) than the higher elevations in the site (3 to 7 cm per decade). Characteristic elevations of salt marsh species and plant communities are developed. The only high elevation marsh communities in the restoration area comparable to those of the controls, were in narrow bands on elevated former dikes.

Without function while diked, former tidal creeks filled in. Upon estuarine reconnection, these vestigial creeks deepened and narrowed under daily pulses of tide. Typically, creeks deepened between 20 and 60 cm over a decade of restoration.

Soils accreted at a mean rate of 3.6 cm per decade as measured in cores taken from sand tags laid on the former pasture surface. Maximum accretion occurred in the lowest elevation areas, particularly in the area dominated by Carex. Although salinities were measured in September (expected maximum), plant species and soil salinity correlated in the expected fashion. We found a significant correlation of Carex with lower salinities in contrast with Salicornia and Distichlis which significantly correlated with with higher salinities. Sandier soils downriver were significantly related to more saline conditions and to the cover of Salicornia and Distichlis.

Function

Although we did not evaluate the usual services supplied by wetlands, we did estimate primary productivity based on species-sorted peak biomass harvest. Estimates of above ground net primary productivity were about the same in the flanking controls as in the pre-restoration diked pasture (1200 $g/m^2/yr$) and comparable to other northwest high salt marshes. Eleven years after dike removal, primary productivity almost doubled in the restored marsh (2300 $g/m^2/yr$). We attribute this increase to the functioning of the low salt marsh plant communities as a rapidly developing ecosystem.

Plant species diversity varied with restoration development. High diversity prevailed in the pasture and diminished rapidly with pasture species mortality; it increased with colonization, and by six years after dike breaching (1984), diversity again decreased as dominant plant communities consolidated the site.

Although a functioning restored salt marsh ecosystem is presently fully integrated into the lower Salmon River estuary, it is not comparable in species composition, structure or function to the original salt marsh that was diked in 1961.

South Shore Study Site

Most of our study was concentrated on the north shore study site where partial dike removal took place in 1978 and where we could assess restoration over an eleven year period. On the south shore, a 63 ha pasture was reconnected to the estuary by dike removal in 1987. Baseline vegetation and elevation data were recorded prior to breaching and site status evaluated one year after dike removal.

Elevations in south shore site were about 15-25 cm lower than in the north river site at the time of dike breaching and about 37 cm lower than the, intact, high marsh controls on the south shore. In 1988, as a result of the lower surface elevations, daily tidal flooding is more intense; much surface water is present; drainage is poor; pasture species were almost completely killed by saline water intrusion shortly after dike breaching, and residual species showed stress. There were few colonizers and probably 80% of the ground cover was bare mud, accumulated plant litter from dead pasture plants, or covered by an algal mat.

We judge the south shore surface elevations barely sufficient to support low salt marsh vegetation. Critical to the rate of recovery on the south shore, will be the development of an

integrated drainage network leading to pulses of tidal water rather than ponding. We established a system of 49 permanent plots along transects, each with a sand-tagged surface marker, and a random system of 688 non-permanent nested frequency plot sets. We intend to continue monitoring recovery of the south shore site.

Prospects

The objectives of the U.S. Forest Service in restoring the Salmon River salt marshes has been to return diked pastures to functioning ecosystems and ultimately to return the estuary to the state prevailing prior to agricultural alteration. Our research demonstrates a successful restoration of a diked pasture on the north shore to a functioning natural salt marsh.

The long term prospects for **complete** restoration to pristine conditions, however, are less clear. Elevation largely controls salt marsh character. Since elevations of the restored marsh are about 35 cm lower than surrounding natural salt marshes, it will take many years for the restored elevations to reach their former levels and for truly natural vegetation to prevail. In the meantime, the surrounding natural salt marshes will change. Tidal connection of the restored marsh and of the flanking, "intact" controls has been altered by excavation and non-functional dikes. Future estuarine character will necessarily reflect a heritage of altered of hydrology. A more realistic restoration goal is the reestablishment of functioning natural salt marsh communities and ecosystems, not pristine conditions.

RECOMMENDATIONS

An overall goal of this research was to provide wetland managers with guidance in planning and monitoring coastal wetland restoration projects in the Pacific Northwest. Of necessity, our recommendations are based on our experience in the Salmon River salt marshes, where diked pastures were returned to tidal wetlands lining the shores of a well-mixed small estuary. Below are a number of general and specific recommendations.

- Restoration of former diked wetlands is more likely to be successful than de novo marsh creation.
- Complete restoration to pristine conditions is probably not a realistic goal. Restoration to a functioning natural system may be a realistic overall goal.
- Explicit statement of restoration goals and means of meeting and evaluating these goals is essential.
- Restoring complete hydrologic connection with the estuary is critical to the success of any restoration.
- On the ground technical supervision of the restoration activity (excavator, bulldozer) by an experienced restoration specialist (agency staff) is essential.
- Hydrologic connection should be made at the position of former tidal creeks, with excavation of creek beds below their former position at the point where tidal creeks cross a dike. This will assure that there will not be a sill inhibiting tidal exchange. Drainage of ebb tide from the restored area is necessary for a successful restoration.
- Complete dike removal is desirable. Although there often is a natural levee between marsh and estuary inhibiting tidal exchange, removal of the entire dike will enhance exchange, including drainage of tidal water from the restored area.
- A carefully conducted elevation survey of the entire area planned for restoration is essential to planning and carrying out the restoration. This survey must be closed and registered to a primary National Geodetic Vertical Datum (NGVD) bench mark. This survey will help predict success and enable planning a planting regime, if necessary. Particular attention should be paid to substantial historic subsidence of the restoration area prior to dike removal.

- The elevation survey must include nearby natural wetlands to determine local elevations of native marsh species. This empirical local information will aid in predicting marsh species establishment. One cannot rely on published regional information concerning the elevation distribution of species with respect to a tidal datum because tidal datum determinations may be erroneous.

- In addition to elevation data, restoration planning should incorporate salinity and soil texture data when choosing planting material, if planting is necessary. Consideration of salinity and texture will also enhance prediction of restoration success.

- Planting of local marsh species will not be necessary in most cases if local propagules of native species are available and if substantial cover of residual native marsh species exists.

- With small restoration projects (an acre or two) and with proper hydrology, salinity and texture, it may be economic to artificially develop appropriate elevations to support desired plant species and communities. With larger restoration projects, artificial elevations will be too expensive to create and it will be necessary to rely on local elevations to determine the nature of the restored wetland. However, caution in needed in designing artificial elevations, as differences of 5 to 15 cm can have a major effects on species establishment. Moreover, artifical substrates often consolidate and subside.

- Post project monitoring should be carried out for at least ten years. There is evidence that the direction and success of the restoration is not apparent until after five years. Monitoring need not be annual, but the monitoring program should involve an objective assessment of recovery and provide for corrective measures if necessary. A "look-see" monitoring program is not satisfactory.

- Specific restoration goals should be used in developing and conducting the monitoring program.

- In monitoring, attention should be paid to: tidal connection, absence of ponding, functioning of tidal creeks, establishment of appropriate wetland species, elevation of established species, and cover of restored vegetation.

- A small system of permanent plots (ca. 1-5 per acre) established over an elevation gradient may facilitate monitoring but is not necessary.

ACKNOWLEDGEMENTS

Our reassessment of the Salmon River salt marsh restoration would never have occurred without the detailed research of Diane Mitchell in 1977-1981. She collected and analyzed the baseline data, supervised its continued collection after completing her degree, and in 1987 helped us relocate all transects and plots, and freely shared her data and ideas.

Most of our field work was carried out in 1988 with the able assistance of Bob Harmon, who also entered and processed mapping data into an automated mapping program. Pete Klingeman, Professor of Civil Engineering at Oregon State University, aided in planning the research and later took charge of the elevation and survey component of the project, being aided in the field by Habib Matim, Eric Kantor, Lara Klingeman, and Derek Klingeman. Ken Bierly, Ken Franklin and John Marshall of the Oregon Divison of State Lands conducted a preliminary elevation survey in 1987, as did staff of the Siuslaw National Forest Hebo District. Dan Luoma assisted in the field and handled biomass sorting, weighing and analysis. We were ably assisted in the field and lab by many individuals including: Tori Bruzese, Peter Eldred, Jerry Griffith, Jim Lenihan, Rosemary Mannix, and Bill Stack. Mark Darienzo was generous in sharing his laboratory techniques for analyzing marsh sediments and we appreciate the use of the Oregon State University Oceanography laboratory facilities. The tedious job of entering data into the computer was carried out by Teresa Brey. Logistic support was provided by Sarah Greene of the U.S. Forest Service Pacific Northwest Forest Experiment Station, Jack Booth, and the staff at Hebo District, Siuslaw National Forest.

INTRODUCTION

A 21 hectare diked pasture in the Salmon River estuary, Lincoln County, Oregon was restored to tidal action in 1978 through partial removal and reconnection of tidal creeks. The restoration site is owned and managed by the U.S. Forest Service as part of the Cascade Head Scenic Research Area. The management direction for the estuarine zone was to reestablish natural salt marsh in those areas where it had been lost. A detailed baseline investigation of the restoration site was conducted from 1978 to 1980 (Mitchell 1981). Our study is a follow-up of the original and evaluates the restoration eleven years after its initiation. Our objectives include describing the status of the restoration after a ten-year period, evaluating restoration processes, and providing guidance for future coastal salt marsh restoration projects in the Pacific Northwest.

Institutional Framework for Wetland Restoration

All coastal salt marshes have been altered to some extent. Although there is much regional variation in the type and severity of alteration, most have been degraded or obliterated entirely (Tiner 1987, Frayer et al. 1983). In the Pacific Northwest, 80-90 % of this loss is accounted for by agricultural diking and draining (Frenkel et al. 1981, Kentula 1986, Boulé and Bierly 1987). Because tidally controlled hydrology is thought to be easily reestablished, diked and drained agricultural lands are viewed as having good prospects for restoration. Accordingly, these agricultural lands have been inventoried in the Pacific

Northwest to assess their restoration potential (Jones and Stokes Associates 1988).

Interest in salt marsh restoration has grown over the past decade as one means of halting or reversing coastal wetland losses (Kusler and Kentula 1989). Most of this interest stems from mitigation policies requiring compensation for unavoidable impacts. An assumption of such policies is that a degraded site can be restored to its pre-impact condition. However, there is more uncertainty than knowledge in such attempts at ecological restoration, as restoration ecology is in its infancy as a science.

The concept that damage to fish and wildlife values could be diminished through mitigation was first introduced in the Fish and Wildlife Coordination Act of 1958 and subsequent amendments (Blomberg 1987, Ashe 1982). The act requires the U.S. Fish and Wildlife Service and state fish and wildlife agencies to cooperatively review development proposals for their effect on estuaries and to determine the means for minimizing or preventing resource damage or loss.

Mitigation acquired a more general meaning and institutional definition in the National Environmental Policy Act of 1969 which addresses the procedure of project analysis and impact mitigation. Among federal agencies playing a role in mitigation decisions are the U.S. Fish and Wildlife Service, National Marine Fisheries Service, Environmental Protection Agency, and U.S. Army Corps of Engineers. In Oregon, four state agencies have a role

in estuarine development and impact mitigation, while in Washington three state agencies play this role. (Blomberg 1987).

Salt Marsh Restoration in the Pacific Northwest

Much of the information on Pacific Northwest salt marsh mitigation and restoration is buried in the records of state and federal agencies responsible for mitigation planning, review, or completion. Except for a recent review by Josselyn et al. (1989), little information is published. The studies cited below are a selection of the major, better-documented restorations of diked salt marshes in the region. A number of reports are also available on marsh restorations in California with major activity in Humboldt Bay (Springer et al. 1984), San Francisco Bay (Josselyn and Buchholz 1984), and San Diego Bay (Zedler 1984). However, the climate and salt marsh resources in California are distinctly different from those of the Pacific Northwest (Macdonald and Barbour 1974, Macdonald 1977) and California restoration projects are therefore not reviewed here.

British Columbia

Most salt marsh studies and restoration projects in British Columbia have been in the Fraser River estuary, which drains a very large area, is relatively fresh, and has been greatly modified by diking (Campbell 1986).

Koksilah Marsh Restoration in the Cowichan River estuary, Vancouver Island, occurs in an estuary where most high marsh has been converted to agriculture (Dawe and Jones 1986, 1987). Restoration of 17.4 ha Koksilah marsh began in 1987 with dike breaching. Pre-restoration data on vegetation and soil-water salinity has been reported (Dawe and Jones 1986, 1987) restoration results are under evaluation.

Englishman River Estuary Restoration was begun in 1979 with the breaching of a 10-year-old dike to restore 40 ha of former salt marsh (Dawe and McIntosh 1987). Vegetation, soil, and elevation data were collected annually from 1979 to 1983 and in 1986. Most upland vegetation was killed within the first year. Areas of bare soil were exposed in the second year together with clusters of newly established salt tolerant species. In the third year, perennial halophytic vegetation increased in cover and frequency. In 1986, vegetation was still changing as indicated by additional salt tolerant species increasing in cover and frequency compared to the more brackish species.

Kokish Marsh Restoration on Vancouver Island differs from others in that historic diking restricted fresh water resulting in a hypersaline marsh (Campbell and Bradfield 1988). In 1985, an 80-year old dike was breached to allow fresh water to re-enter the marsh with the expectation that with diminished soil salinity, productive Carex lyngbyei would dominate. Pre-restoration vegetation, soils, and elevation data were collected together with 15 month's post-restoration data. A nearby undiked

marsh served as control. Dike breaching success was less than anticipated as river water only entered the site at high tide. Soil salinity did not appreciably decrease over the 15-month period, remaining twice as high as that of the control site. Carex lyngbyei has not reestablished.

Washington

Cooper (1987) reviewed 18 mitigation projects in the state from 1983 to 1987 from a planning and management perspective. Two of the projects aiming to restore former salt marsh received some monitoring. Cooper notes that the effectiveness of most habitat mitigation projects is either unknown or incompletely documented.

Lincoln Street Marsh Restoration in the Puyallup River estuary was intended to compensate for the Port of Tacoma fill activity and approaches being a marsh creation project (Strickland 1986; Cooper 1987). Design characteristics include "fingers" of shoreline to increase marsh length and enhance juvenile salmonid habitat, creation of mud flat, cattail marsh, grassland, and shrub forest habitats. Construction included grading the site to its former elevation, building a new dike, and breaching a connection to the river. Surveys shortly after construction revealed some erratic depths rather than the gradual gradient planned; however, there was good initial survival of Carex lyngbyei, 24 species of birds, and 11 species of fish, including juvenile salmonids.

Oregon

Despite the potential for diked salt marsh restoration in Oregon, few strict restorations have been undertaken. Most marsh compensation for mitigation has been creation, i.e., converting upland to intertidal marsh (Kentula 1986, Fishman Environmental Services 1987). The Salmon River restoration is the only project not related to mitigation for development.

Coalbank Slough Mitigation is aimed at restoring a 2.4 ha upland pasture in Coos Bay to former high salt marsh. Two 31 m openings were made at MLLW in a 595-meter-long dike in 1987. No post-breach evaluation has been reported. The permit requires that the site be monitored for five years, but contains no specifics as to what is to be monitored or how monitoring is to be conducted. No pre-mitigation survey was undertaken.

Astoria Mitigation Bank is on a 14.4 ha site adjacent to Young's Bay on the Columbia River and is the state's first (and presently only) mitigation bank. The area was diked for pasture in the late 1890's and, in 1985, supported pasture vegetation, shrubs, and small trees. In the winter of 1986/1987, a new dike to contain the site was constructed and five small openings made in the old dike to reconnect creeks. Limited floristic and hydrological baseline data was collected in 1985 and 1986 and post-breach data was collected annually since (Jackson 1987). Hydrological problems were immediately apparent, including restricted tidal circulation and establishment of a freshwater wetland with standing water of 15 to 61 cm over much of the site.

No pre-restoration engineering analysis was done and it appears that continued hydrologic problems prevent restoration to intertidal brackish marsh.

Restoration Problems and Needs

Mitigation-related restoration "...tests the present limits of understanding of estuarine environments" (Blomberg 1987). Due to a lack of knowledge about wetland ecology, restoration projects are undertaken with little idea as to whether or not the project is replacing lost habitats, functions, and values. Expectations are often vague. Few solid guidelines are given for design, execution, monitoring or evaluation of results. Areas of uncertainty are poor identification of environmental factors influencing restoration results, absence of design criteria, use of inappropriate techniques for achieving predictable results, and lack of monitoring guidelines (Cooper 1987, Good 1987, Josselyn et al. 1987, Kusler and Kentula 1989, Race 1983, 1985, Zedler 1983, 1986, Strickland 1986). Among problems are: local ponding; failure to obtain original vegetation due to subsidence, improper slope and tidal elevations; poor tidal circulation; poor tidal channel development; poor or no vegetation regeneration; and failure of plantings.

To redress some of these problems, a number of articles and manuals providing guidelines for design, execution, and monitoring of salt marsh restoration projects have appeared over the past decade (Haltiner and Williams 1987, Harvey et al. 1982, Josselyn 1982, Josselyn and Buchholz 1984, Kusler and Kentula

1989, Smith 1983, Williams and Harvey 1984, Zedler 1984). Electronic databases and bibliographies including restoration and mitigation topics are also improving information availability and exchange (Council of State Governments, U.S. Environmental Protection Agency, Wolf et al. 1986). Although data bases may be important, critical aspects of salt marsh restoration have rarely been studied and are poorly understood, so agency managers and planners often operate "... in an information vacuum" (Strickland 1986).

Correction of many of these problems has been slow because few long term successful restoration examples exist, restoration projects are generally very small and poorly documented, few studies include controls for comparisons, and evaluations are often vague, qualitative (not quantitative) and misleading.

Study Objectives

We address many of the above needs by evaluating in detail a relatively large (21 ha) restoration site over a fairly long term of eleven years. Moreover, by using quantitative data we compare restoration site status at year eleven with the status of adjacent relatively intact control sites. The objective of the restoration is clearly stated; namely, the rehabilitation of the estuarine wetland to its condition prior to diking and agricultural use. We identify three characteristics by which restoration can be evaluated, the **composition** of the restored salt marsh, the environmental **structure** of the site, and the **function** of the wetland ecosystem.

Composition

Assessment of vegetative composition was conducted by applying similar data collection and analytical procedures to a set of permanent plots which have been maintained over the eleven-year restoration period. Trends in species composition, cover, and plant communities were examined in the restored and control areas. Although distribution of species and communities may be regarded as a structural property of the wetland, we treat this as a compositional characteristic.

Structure

By structure we mean the spatial arrangement of environmental factors which control species and community distribution in the restoration area. This conceptual approach considers the role of certain key environmental characteristics: (1) elevation (a major control on the restoration); (2) the special structural element of leveled dikes; (3) changes in creek morphology (a control on tidal exchange); (4) sedimentation and accretion of inorganic and organic sediments; (5) substrate texture; and (6) salinity.

Function

Wetland functions are usually considered as the services provided by wetlands, such as fish and wildlife habitat, flood control, water quality enhancement, and amenity values. Besides these wetland functional values which we did not evaluate are ecosystem functions. Three functional properties of ecosystems which are often considered are: energy flow, nutrient cycling,

and diversity. As a measure of energy flow, we estimate productivity from species-specific peak biomass. We collected no data on nutrient cycling. We apply simple diversity measures.

Restoration Monitoring

Most restoration projects receive cursory monitoring at best (Kusler and Kentula 1989). Yet restoration success and the ability to apply corrective measures to a poorly designed restoration project are dependent on a sound monitoring system. An additional objective of our research was to test an alternative vegetative monitoring approach to that usually applied.

Permanent plots are often established to assess vegetation change over time. As discovered by many researchers, we found that maintenance of a permanent plot system over a period of a decade or more is very difficult. This is particularly true in a dynamic environment such as a salt marsh where tidal-born debris wipes out permanent stakes and can obliterate transects. We have attempted to evaluate a new system of vegetation trend sampling using the nested frequency method which provides a statistical assessment of vegetation change. In doing this, we applied the technique to a newly restored marsh; however, we will be unable to make a full evaluation of this technique for a number of years.

SALMON RIVER STUDY AREA

Reestablishment of the Salmon River estuarine salt marshes has been purely a restoration program, not a mitigation action. Undertaken by the U.S. Forest Service in 1978, dikes enclosing two pastures on the north shore of the estuary were breached and the pastures reconnected to tidal circulation in order to reestablish a natural salt marsh without any manipulation other than dike removal and tidal creek connection. Subsequently, dikes on the south shore of the estuary were removed in 1987 to return 63 hectares of diked pasture to marsh. In this section we discuss the institutional framework for these actions and details of the restoration history.

Institutional Setting

In 1974, Congress designated 3400 ha in the Siuslaw National Forest, including the Salmon River estuary, as the Cascade Head Scenic-Research Area (CHSRA). The purpose of this dedication is to: "... provide present and future generations with the use and enjoyment of certain ocean headlands, rivers, streams, estuaries, and forested areas for research and scientific purposes, and to promote a more sensitive relationship between man and his adjacent environment" (U.S. Laws, Statutes, etc. Public Law 93-535, 1974).

The CHSRA was subdivided into a number of zones. In the Estuary and Associated Wetlands Subarea, the long-term goal 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. It should be rehabilitated to its condition prior to the existing diking and agricultural use" (USDA Forest Service 1976). To achieve this goal, the Forest Service developed a primary management objective for the subarea permitting the breaching of existing dikes.

Salmon River Estuary and Restoration Action

The Salmon River estuary (Figure 1), the second smallest in Oregon, encompasses about 800 ha. The 40.7 km river drains about 194 sq km and has an annual discharge of 5.5×10^8 m^3 and an estimated annual sediment load of 12,700 m tons/km^2 (Percy et al. 1974). Above river km 3.0, sediment is of fluvial origin; from km 3.0 to 2.0, fluvial and marine sediments are mixed; below km 2.0, marine sediments dominate (Peterson et al., 1984). Prior to the early 1960's, most of the Salmon River marshes were undiked but frequently used for pasture and haying. From 1961 to 1963 approximately 254 hectares of high marsh were removed from tidal connection by dikes fitted with tide gates on both the north and south shore of the river. The status of Salmon River estuarine lands in 1978 is summarized in Figure 2. In 1978, approximately 65 % of the estuarine lands had been converted to agriculture or other uses and 27.5 % of the original high salt marsh remained undiked and relatively intact.

In accordance with the CHSRA goal of restoring the estuary, in 1978 the Forest Service acquired two diked pastures enclosing 22 ha on the north shore of the Salmon River. About half of the

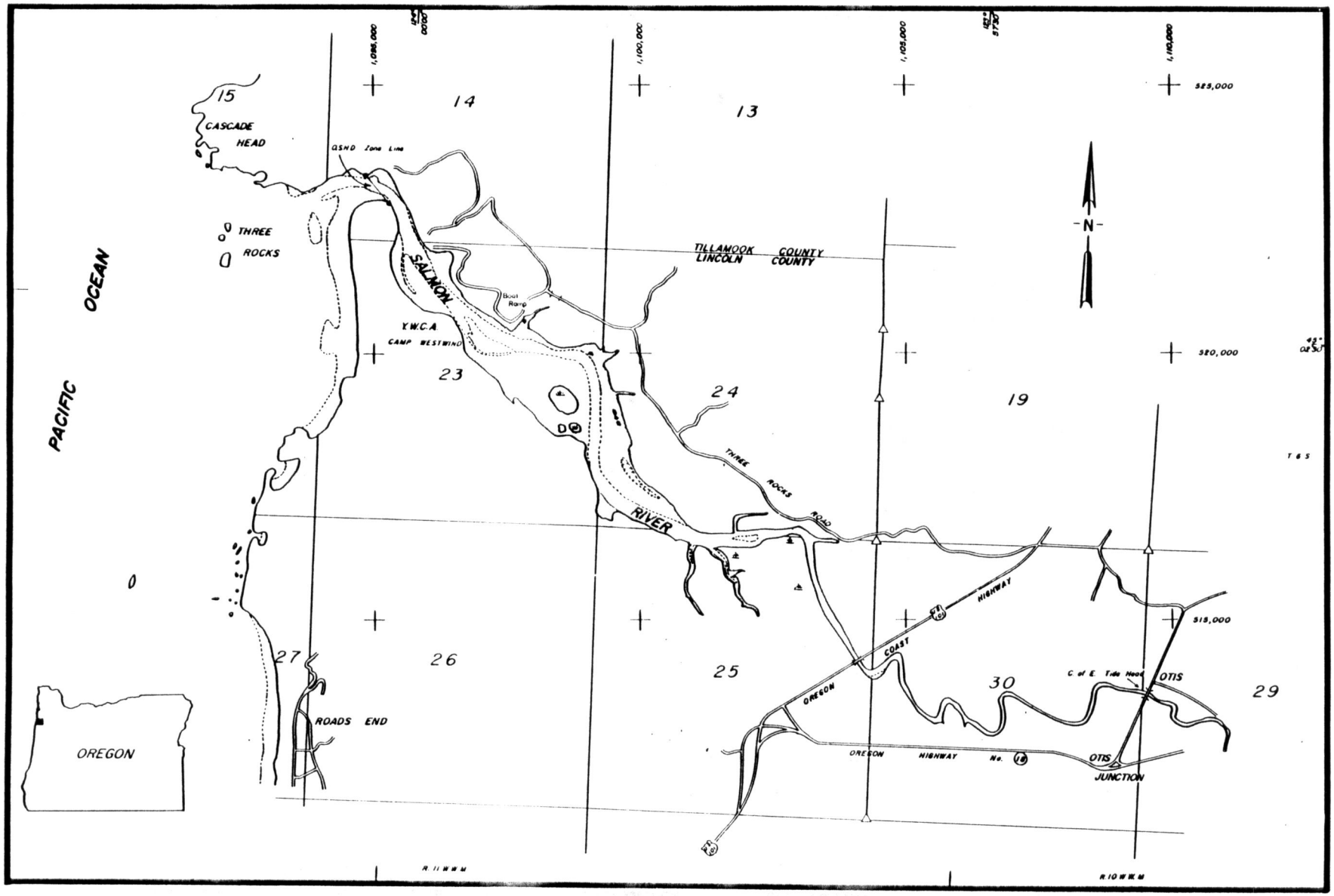


Figure 1. Location of Salmon River estuary, Oregon (after: Division of State Lands, 1973)

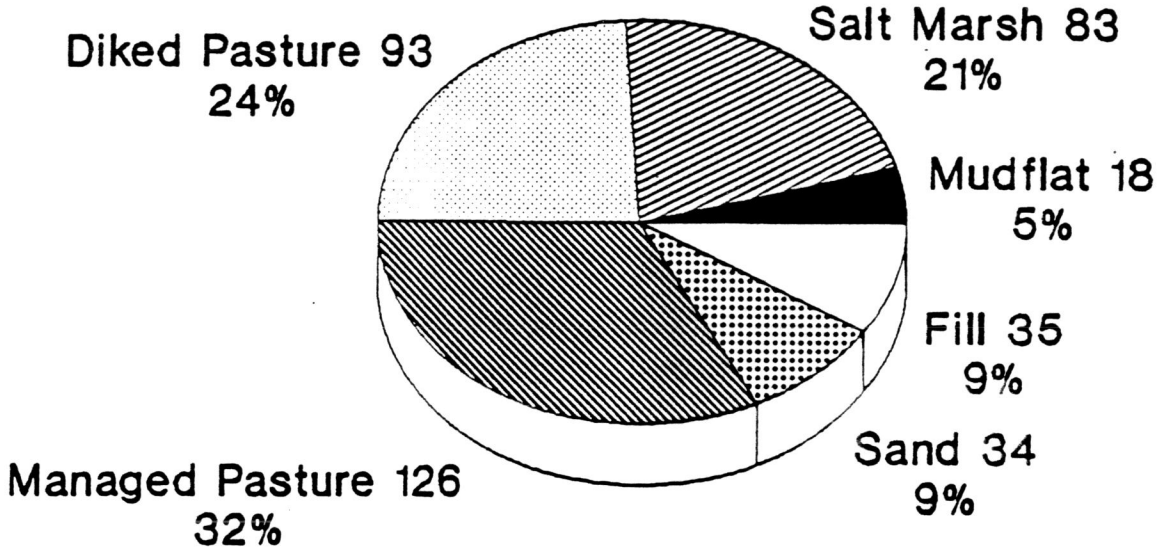


Figure 2. Distribution of land use types in the Salmon River estuary in 1978 prior to restoration actions. Areas are give by percent and hectares.

1550 m outer dike was removed by two small bulldozers in late summer of that year. Removed material was deposited in the interior ditch formed in 1963 when the dike was built. Major creek connections were reestablished by backhoe excavation. Prior to and coincident with dike-removal, the Forest Service funded Diane L. Mitchell, an Oregon State University Ph.D. student, to study salt marsh reestablishment in the pasture area. The present study is a continuation of Mitchell's (1981) research.

Continuing with the long-term goal of restoring the Salmon River estuary, the Forest Service acquired a 63 ha diked pasture on the south shore of the river in 1985. In late summer 1987, all of the dike enclosing this pasture was removed by excavator and bulldozer. Additionally, except for a small stub of dike, the remaining dike on the north shore of the river was levelled at the same time.

Study Sites and Study Areas

We assess two study sites, one on the north shore of the Salmon River and the other directly opposite on the south shore (Figure 3). The north shore study site includes the 1978 restoration area and two flanking "controls" studied by Mitchell. The south shore study site includes the large area (Area 57) which, in 1987, was returned to tidal circulation by dike breaching. For clarity and consistency we continue Mitchell's (1981) labeling of sub-study areas as shown in Figure 3 and listed below.

Area	Location	Treatment
50	North shore	Control area
51	North shore	Restored 1978
52	North shore	Restored 1978
53	North shore	Control area
54	South shore	Control strip
55	South shore	Control area
56	Estuary center	Control island
57	South shore	Breached 1987

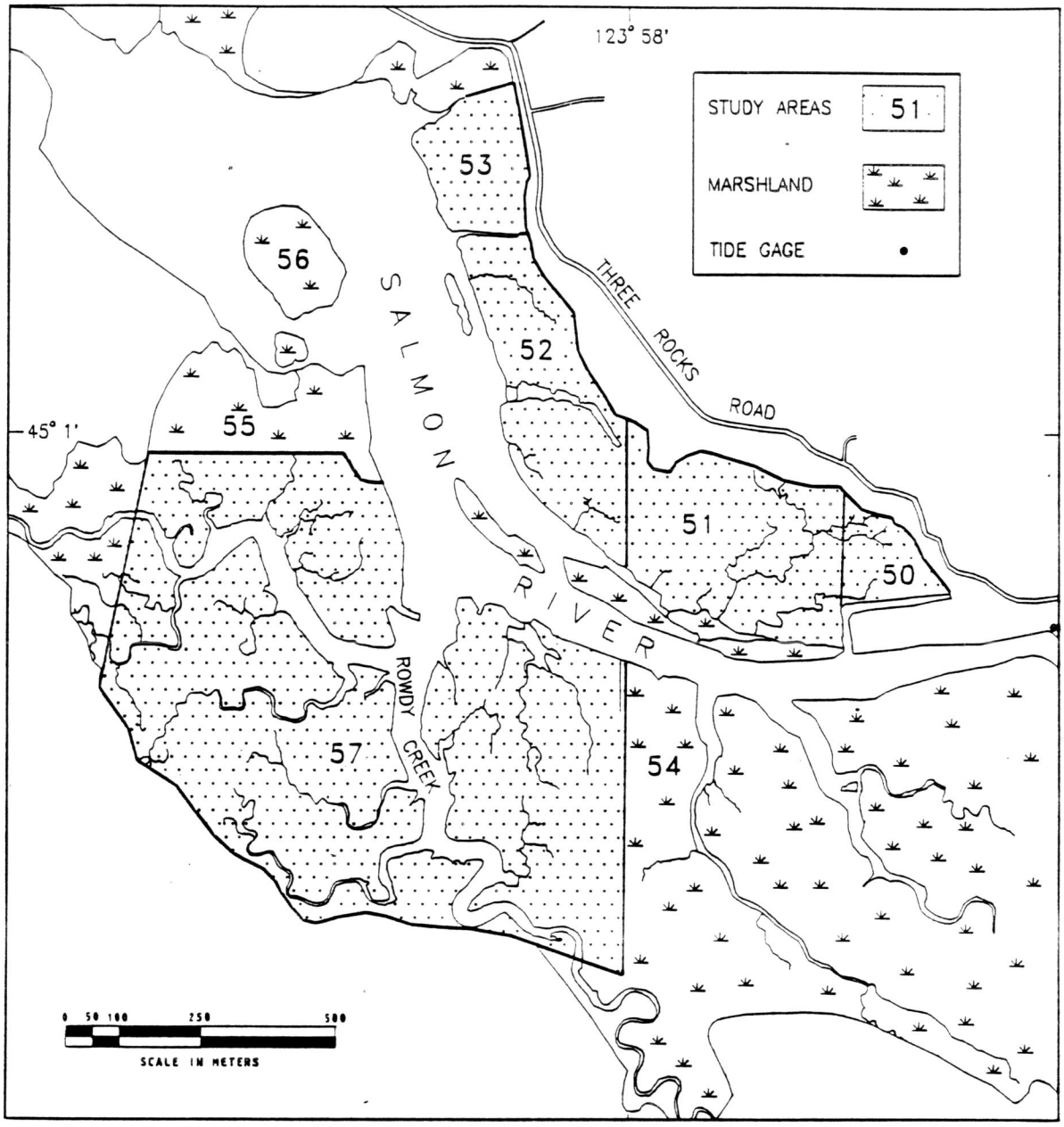


Figure 3. North and south shore study sites in the Salmon River estuary showing restoration areas (51, 52, and 57) and control areas (50, 53, 54, 55, and 56). Dot on right edge of map locates tide gage site.

According to Mitchell (1981), Areas 51 and 52, separated by a property line, were diked in 1963. In 1977, Area 51 was grazed by approximately 15 head of cattle, continuing a long trend of cattle use, while Area 52 provided pasture for two horses. In the early 1960's, six horses were pastured in Area 52. The diked pastures were connected to the river by a single tide gate at the mouth of the large creek in Area 52. A ditch bounded the upland margin of Area 51 and drained into the creek with the tide gate. No other draining, plowing, or seeding took place.

The north shore "controls", Area 50 and 53, were assumed by Mitchell (1981, p.15) to be "fragments of relatively intact high salt marsh". In fact, tidal connection in each was much altered. Area 50 was isolated from some of it's original drainage by the dike separating it from Area 51, and, by the early 1960's, construction of an abortive marina channel. Area 53 was partially diked in the early 1950's, but the dike soon eroded and was never rebuilt (Mitchell 1981). None-the-less, there is presently an abrupt and narrow creek connection on the south end of Area 53 and a broad connection to north. Much of Area 53 is flooded at only the highest tides.

South shore Area 57 was diked in 1961-2 and fitted with a tide gate at the mouth of Rowdy Creek. This 63 ha high salt marsh had been mowed for hay prior to and after diking, grazed by cattle and more recently by horses. Areas 54 and 55 flank Area 57 and support relatively intact high marsh. Drainage in each was effected by the ditch created by removal of dike material. Area 56 is a small island with low salt marsh.

Mitchell Research

Mitchell's (1981) research objective was "... to evaluate the potential for natural salt marsh reestablishment after dike breaching...." To accomplish this task, she (1) established permanent plots along transects in the restoration area and intact "controls"; (2) described pre-restoration plant communities, soils, and elevation on and off the restoration site before breaching in 1978; (3) breached the dike and reconnected tidal creeks with the estuary; (4) resampled vegetation and selected environmental measures for two years (1979, 1980) after breaching. She also collected, but did not report, vegetation data collected in 1981, 1982 and 1984.

Prior to dike breaching, three pasture communities covered Areas 51 and 52: the Potentilla pacifica community, the Agrostis alba-Holcus lanatus community, and the Holcus lanatus community. Elevations among communities did not differ significantly and Mitchell attributed the spatial pattern, in part, to different agricultural practices (horse vs. cattle). Creeks exhibited broad, shallow, vegetated cross-sections in which an Agrostis alba-Eleocharis palustris plant community dominated.

After breaching and salt water inundation, pasture plant communities changed completely, creeks flooded and their cross-sections deepened. Mitchell allocated species into three "types": upland -- present in 1978, dead in 1979; residual -- present in 1978, persistent in 1979 and 1980; and colonizing -- absent in 1978, new in 1979 or 1980. Control Areas 50 and 53 showed little change in species composition. In the restored

area upland species, initially comprising 90 % cover, declined dramatically. Residual species declined from 80 to 40 % cover. After breaching, colonizing species increased from 15 to 21 % cover. "Bare ground" increased from 0 to 45 % surface cover. Biomass shows the same trend as did relative cover. By 1988, three colonizing species (Carex lyngbyei, Salicornia virginica, Distichlis spicata) comprised over 70 % of the vegetative cover in the restoration area. Two other species with appreciable cover were residual (Potentilla pacifica, Agrostis alba).

Mitchell also traced percent cover, frequency and biomass of individual species over the three year restoration period. Of significance to the present study, is that a number of important colonizers in the first two years after breaching vanished by 1988. These include: Cotula coronopifolia, Spergularia marina, and Puccinellia pumila.

In investigating vegetation response to inundation, Mitchell grouped permanent plots into four "persistence classes" reflecting the amount of exposed bare ground. Persistence Class I plots had little bare ground and no new colonizers in the two years after breaching, i.e., they were fully covered by residuals. Plots in this class had the highest elevation and were least effected by flooding. Plots in Persistence Class IV had less than 75 % vegetative cover in 1979 and, being lower, were most effected by flooding. These plots were most subject to colonization. According to Mitchell (1981), "by 1980, in areas of high residual species cover (P.C. I and II), the most successful colonizers were Atriplex patula,...Hordeum

brachyantherum,...and Carex lyngbyei.... In P.C. III and IV areas, where there was mostly bare mud, Spergularia marina...and Salicornia virginica...have colonized most rapidly." From this analysis, Mitchell demonstrated the importance of elevation differences as small as a few centimeters (5 to 15 cm) to species composition. These same small elevation differences still exert control on the vegetation of the restored area.

On leveled dikes, Mitchell identified four vegetation types: Spergularia Type on surface elevations of 1.10-1.20 m NGVD; Deschampsia Type at elevations of 1.35-1.45 m NGVD; Bare Ground Type on compacted dry substrate on slightly higher elevations than the Deschampsia Type; and Upland Type with alien pasture species at elevations of 1.55-1.70 m NGVD. By 1988, these leveled dike vegetation types were still recognizable with the exception of the Spergularia Type which was replaced either by Carex lyngbyei or Salicornia virginica-Distichlis spicata.

After dike breaching, tidal creek vegetation was killed by flooding, and bank vegetation reflected the surrounding marsh surface communities. Mitchell detected slight deepening of creek cross-sections (< 1 cm) one year after breaching.

Mitchell measured 0.30-0.40 m subsidence of the diked site (diked in 1961-63) relative to the intact "controls" which supported higher transitional marsh types. Elevation of the subsided restoration area was at a level at which she anticipated development of an "... intertidal to low transitional salt marsh." Eleven years later, the restored site supports low intertidal marsh. Her prediction has proven to be accurate.

METHODS

Field Methods: North Shore Study Site

Transect System and Permanent Plots

Mitchell (1981) established a system of permanent plots to facilitate sampling over a period of years in the north shore study site (Figure 4). In 1978, she laid out 12 transects oriented perpendicularly to the Salmon River along which she placed 49 permanent plots, the first of which was located within 10 m of the upland. Subsequent plots were spaced at 25 to 45 m intervals, depending on topography. Three transects with 11 plots were established in control Area 50, and in 1979, 24 plots were located along five transects in control Area 53. Additionally, in 1979 Mitchell placed 23 permanent plots along the distal end of transects on leveled dikes. About one-third of the original dike remained intact and was finally leveled in 1987. Therefore, in 1988 we established 17 new permanent plots on the recently leveled dike. The entire north shore study site now includes 20 transects and 115 permanent plots.

In 1988, we located and remarked Mitchell's original plots. Each 3x3 m plot was identified at the NW (primary) corner by a 6-foot length of 3/4 inch PVC pipe. The other three corners were marked with short cedar stakes. The most northern transect in control Area 53 was abandoned in 1988, leaving 4 transects with 29 plots in that area; all other plots were retained. Data gathered from permanent plots in the restoration area includes:

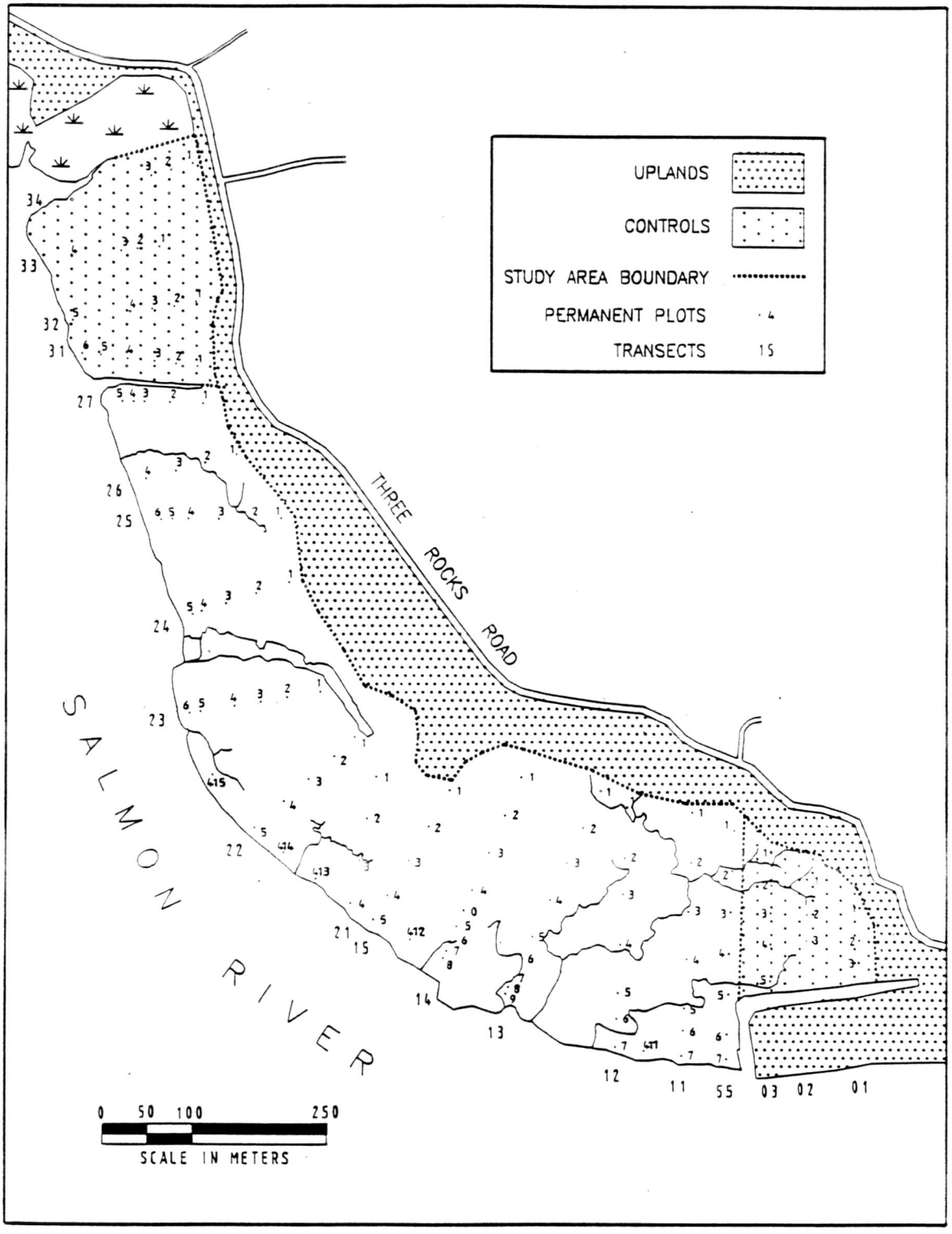


Figure 4. Numbered permanent plots set out along numbered transects in the Salmon River estuary north shore study site.

1) percent cover of all vascular plant species;
2) above-ground biomass sorted by species;
3) photograph (color transparency) of 1x1 m plot at the primary corner where vegetation cover was sampled;
4) interstitial soil-water salinity;
5) soil texture;
6) organic content of soil;
7) accretion above 1978 sand-tagged surface;
8) location within transit survey network; and
9) elevation.

Information from control area plots includes:

1) percent cover of all vascular plant species;
2) above-ground biomass of all vascular plant species;
3) location within transect survey network; and
4) elevation.

Vegetation Sampling

Cover. Canopy cover classes (Table 1) of all vascular plant species were recorded for a 1x1 m plot at the primary corner of each permanent plot. Cover data was collected by Mitchell in mid-late August 1978, 1979, 1980, 1981, 1982, and 1984. We sampled cover between 29 June and 7 July, 1988. A 35 mm color slide was taken of each 1x1 m plot to show vegetative cover changes over the restoration period. A step ladder permitted a near-vertical camera position above the plot.

Creeks. Mitchell described vegetation across 47 creek cross-sections in 1978 and 1979 (Figure 5). We resurveyed vegetation cover along these same transects at selected intervals reflecting major vegetation change. We used a 20x50 cm sampling frame, the short axis placed parallel with the cross-section and on the up-stream side. Cover class was recorded for each species (Table 1).

Table 1. Cover classes used for vegetation data collection.

Cover Class	Percent Cover Range	Percent Mid-point Cover
p	present outside plot	-
+	< 1	0.5
1	1 - 4.9	2.5
2	5 - 9.9	7.5
3	10 - 24.9	17.5
4	25 - 49.9	37.5
5	50 - 74.9	62.5
6	75 - 94.9	85.0
7	> 95	97.5

Biomass. Above ground biomass was sampled in the SW corner of each permanent plot in the restoration area by clipping all vascular plants at ground level within a 20x50 cm sampling frame. Cover class of each species was recorded before clipping, and unsorted green sample weight recorded on site. Biomass was sampled in 49 plots by Mitchell in the restoration site July-August 1978, 1979, 1980, 1981, 1982, and 1984. We clipped biomass on 25 July 1988, close to the same dates used by Mitchell. Eleven control site plots were harvested by Mitchell along the two transects adjacent to the restored site in 1978 (5 plots in Area 50 only), 1979, 1980, 1981, 1982, and 1984. We did not sample biomass in the control areas in 1988. Because of vegetation uniformity in Area 51, in 1988 we took a 30 percent

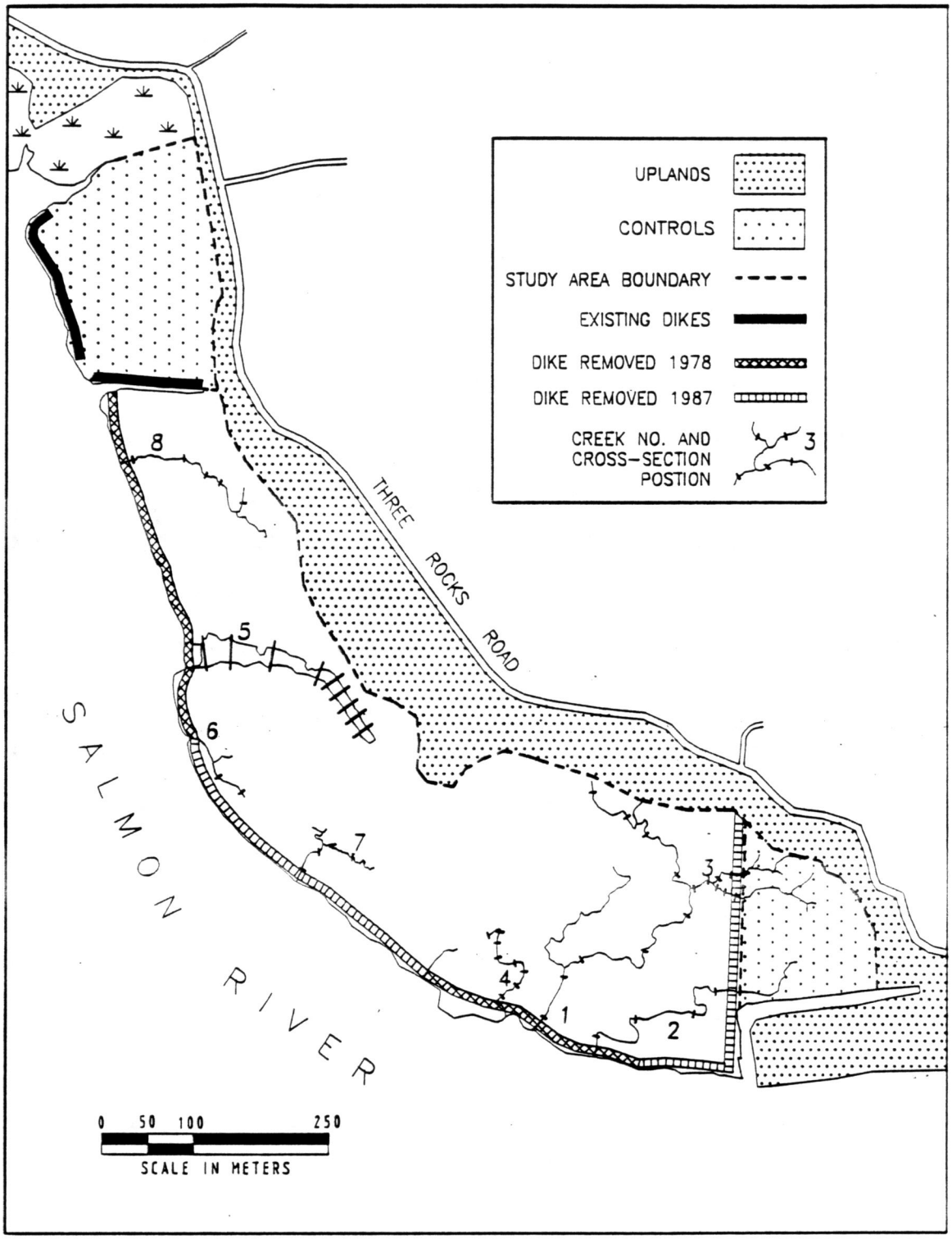


Figure 5. Numbered creeks, creek cross-section positions, and dike removal history in the Salmon River estuary north shore study site.

subsample of all plots dominated by Carex lyngbyei (cover greater than or equal to 75 % cover and other species less than 1 % cover). Subsample selections were based, in turn, on stratification of samples into four vigor classes based on Carex height: 90-110 cm, 110-130 cm, 131-150 cm, and 150-170 cm.

Elevation and Mapping

Previous elevation surveys. In 1978, Mitchell surveyed elevations in the north shore study site with an automatic level, excluding Area 53. Elevations were recorded at primary corners of permanent plots and at 5 m intervals along transects. Her survey was registered to three temporary bench marks (TBM) adjacent to Three Rocks Road (two marked rock points, one on a log). She also surveyed elevations of all creek cross-sections at intervals of approximately 20 cm, which were tied into the elevation net. Creek cross-sections were resurveyed in 1979 and 1980. In 1979, she resurveyed the study site, including Area 53, carrying the survey across to the south shore of the Salmon River, thence east, back across the river to the 7.00 foot mark on the tidal staff gage near the Jack Booth residence. Henry Pittock, later in 1979, tied the 7.00 foot tidal staff elevation to two temporary bench marks on the Jack Booth property. Mitchell also tied her elevation net into a survey being conducted 9 October 1980 by the U.S. Forest Service Regional Office which, in turn, was registered to a USCGS "...bench mark designated B667 (1975) located at the southwest corner of the overpass bridge of State Highway 18 over U.S. Highway 101..."

(Mitchell 1981). However, the tie-in to the U.S. Forest Service survey proved inaccurate and was abandoned in our reassessment.

Although Mitchell reported all elevations relative to the MLLW tidal datum and, by reference to the U.S. Forest Service survey could express elevations with respect to NGVD, careful examination of Mitchell's field notebook showed that she never properly closed her elevation network. It was not until 1988 that the net was closed to 4 mm!

1988 elevation survey. We conducted a preliminary survey in 22 September 1987 with the help of the Oregon Division of State Lands wetlands staff to determine roughly whether elevations changed in the study site. Apparently they had.

A third-order survey was run in mid-July 1988 including the north and south shore study sites, previous temporary bench marks established by Mitchell and Pittock, new bench marks established in the present study, and the NOAA-NOS primary National Geodetic Vertical Datum (NGVD) Bench Mark R-713 at the Pacific Coast Trail parking lot at Three Rocks Road and U.S. Highway 101. All elevations in the present study are referenced to NGVD, which is the most stable local datum. Tidal data collected by Mitchell for 16 months is available to reference elevations to the MLLW tidal datum where appropriate. Bench mark elevations are listed in Appendix 1.

Five elevations were measured at each permanent plot: four surface elevations at plot corners and a TBM elevation on top of the SW corner cedar stake. A number of other TBM's were set up

to facilitate future resurveying. Four measurements per plot allowed us to determine surface roughness. Repeat measurements comparing Mitchell's 1978-1979 elevations with our 1988 elevations were taken at the same point (northwest corner of plot).

Creek cross-section elevations were resurveyed at 46 relocated positions, with care in placing the stadia rod in soft sediments. Elevations were tied to NGVD.

Field mapping. The entire study site was field-mapped with transit and stadia to register angles and distances between fixed topographic points and transect ends. To aid in mapping, six aluminum aerial photo targets were placed in the north and south shore sites and incorporated into the mapped network; however, attempts to have imagery flown were unsucessful. Mapping data permitted preparation of accurate maps with the aid of the mapping programs, SURFER and AUTOCAD.

Soils and Accretion

In 1988, data was collected for interstitial soil-water salinity, soil texture, soil organic content, and accretion. Two soil cores (3.4 cm diam. x 15 cm long) were extracted from each restoration site permanent plot. Soil core 1, extracted from the biomass collection plot, was used to measure soil-water salinity and was retained for analysis of soil texture as described under laboratory analysis, below. Soil core 2, inserted into sand tags laid down in 1979 in the northeast corner of the permanent plots,

was used to measure accretion after dike breaching, and was retained for laboratory analysis of organic content.

Soil-water salinity. At the SW corner of each permanent plot, a 1 cc soil sample was cut from soil core 1 at 5 cm depth corresponding to the main root zone. The soil salinity sample was wrapped in hardened filter paper and placed in a disposable syringe. A few drops of clear interstitial water were expressed onto an American Optical hand refractometer, and salinity read in the field in parts per thousand on 7-9 Sept. 1988.

Accretion. In 1978, a sand tag was placed by Mitchell in the NE corner of each permanent plot. Her objective was to mark the marsh surface existing at that time, allowing future measurements to be taken of accreted marsh surface above the sand tag. Tags measured 25x25 cm and were approximately 1.5 cm thick. To facilitate relocation, the sand-tagged corner was marked by Mitchell with a cedar stake.

In 1988, we cored into the center of each tag which could be relocated. Of the original 49 tags in the restoration site, 44 were located. The coring device was a section of chromium-plated brass tubing (3.4 cm diam. x 40 cm) sharpened at one end and threaded at the other into a drain pipe T-joint serving as a handle. The device was inserted with a slow, twisting motion to a maximum depth of 15 cm, and the soil core was extruded from the pipe from the core bottom to minimize core compaction above the sand tag. The core was extruded into a cradle, loose, undecomposed litter brushed from the top, and the distance from

the soil surface to the top of the sand tag was measured to the nearest millimeter. The core was wrapped in plastic and retained in a refrigerator for further laboratory analysis.

Laboratory and Analytical Methods: North Shore Study Site

Vegetation

Floristic data was analyzed by three multivariate programs: two-way indicator species analysis or TWINSPAN, a classification program (Hill 1979a); a clustering routine in the PC-ORD system, also a classification program (McCune 1987); and detrended correspondence analysis or DECORANA, a modified ordination program (Hill 1979b). The results from these programs generally agreed with each other. Diversity was measured using PRHILL, an unpublished computer program developed by B.G. Smith which calculates jackknife estimates of Hill's diversity numbers (Hill 1973) for each community. All statistical analyses was performed by STATGRAPHICS, a general purpose statistical package.

Biomass was handled in the laboratory. Each sample was sorted to species, including the current year's dead biomass production. Litter was recorded separately. Sorted samples were placed in drying ovens at 100 C, weighed and reweighed until dry, the time varying by species. Dry weights for each species per sample were recorded to the nearest gram. Productivity was calculated by the method of Kibby et al. (1980) by which the individual seasonal dry weight relations for dominant species are used to estimate community primary productivity.

Soils

Soil cores collected from each permanent plot were analyzed for soil texture (grain size) and organic content (percent dry weight loss on ignition). Samples from each core were collected from the top 2 cm of the core in order to analyze soil material accreted after dike breaching. Sand tag cores (where available) were used for organic content analysis. A second core was extracted and used for grain size analysis in order to avoid polluting samples with introduced sand used to tag surfaces. Methods detailed below approximate those refined by Darienzo (1987).

Soil texture. Samples were placed into a 600 ml beaker and weighed. Distilled water was added and samples mashed with a rubber policeman to remove lumps. Gross plant material (root and stem fragments) was removed, with attached soil rinsed back into the beaker.

Samples were placed on a vented hot plate at 93 C and treated with 20 ml of 30% hydrogen peroxide to further reduce soil clumping and dissolve organic material. Samples were stirred occasionally until most frothing ceased (about 1 hr) and were then set aside overnight.

The following day, samples were wet-sieved through a series of three sieves to remove remaining coarse organics and separate the sand component from the silt/clay component. A 991 micron sieve removed coarse organics, a 246 micron sieve isolated medium

sands, and a 62 micron sieve retained fine sands, allowing silt and clay to pass into a large collection jar. Material from the latter sieve was repeatedly mashed with a rubber policeman to break down small clay clumps, and resieved.

The sand fraction was transferred from the sieve to a small beaker, excess organic scum was floated off the top, and the sample was oven-dried at 60 C. The silt/clay fraction in the collection jar was candle-filtered to remove excess water from sieving process and transferred to a beaker to oven-dry overnight. Dried samples were weighed on a Mettler balance to nearest mg and percentages of sand and silt/clay calculated.

Organic content. Samples were collected from the top 2 cm of soil cores, avoiding surface litter and large roots, and placed in crucibles in which they were oven-dried at 100 C for 24 hours, then transferred to a dessicator to cool. Dried samples were weighed, fired in a furnace for 15 hours at 380 C, cooled again in a dessicator, and re-weighed. Percent organic content was inferred from percent loss on ignition (LOI) determined by:

$$\text{LOI } (\%) = \frac{W_1 - W_2}{W_1} \times 100$$

where: W_1 = oven dry weight precombustion
W_2 = oven dry weight postcombustion

South Shore Study Site

Dike Removal

In 1987, the south shore study site consisted of Area 57, a 63 ha pasture that was diked in 1962, and two undiked, relatively intact natural salt marshes in Area 55 and 56 (Figure 3). The U.S. Forest Service acquired this area in 1985 with the intention of returning the pasture to natural salt marsh by dike removal. Dike leveling was conducted in late August 1987 by an excavator, bulldozer and two dump trucks. Dike material on the north and east edge of Area 57 was placed in the **ditch** that was between the east dike and Transect C (Figure 6). Also, excess material from the outer dike fronting the Salmon River was dumped into this ditch. The reason for this disposal was that, upon diking in 1962, Rowdy Creek became diverted into the east dike ditch and subsequently this channel deepened due to daily and seasonal tidal erosion and freshwater runoff.

1987 Field Methods

Because of time limitations prior to dike removal in 1987, different field methods were employed on the south shore study site than on the north shore. Although visited by Mitchell in 1978-1980, much of the south shore study site was not surveyed in detail and no permanent plots were established. Mitchell did conduct a vegetation survey in natural (control) salt marsh along transects in Area 54, 55, and 56 (Figure 3).

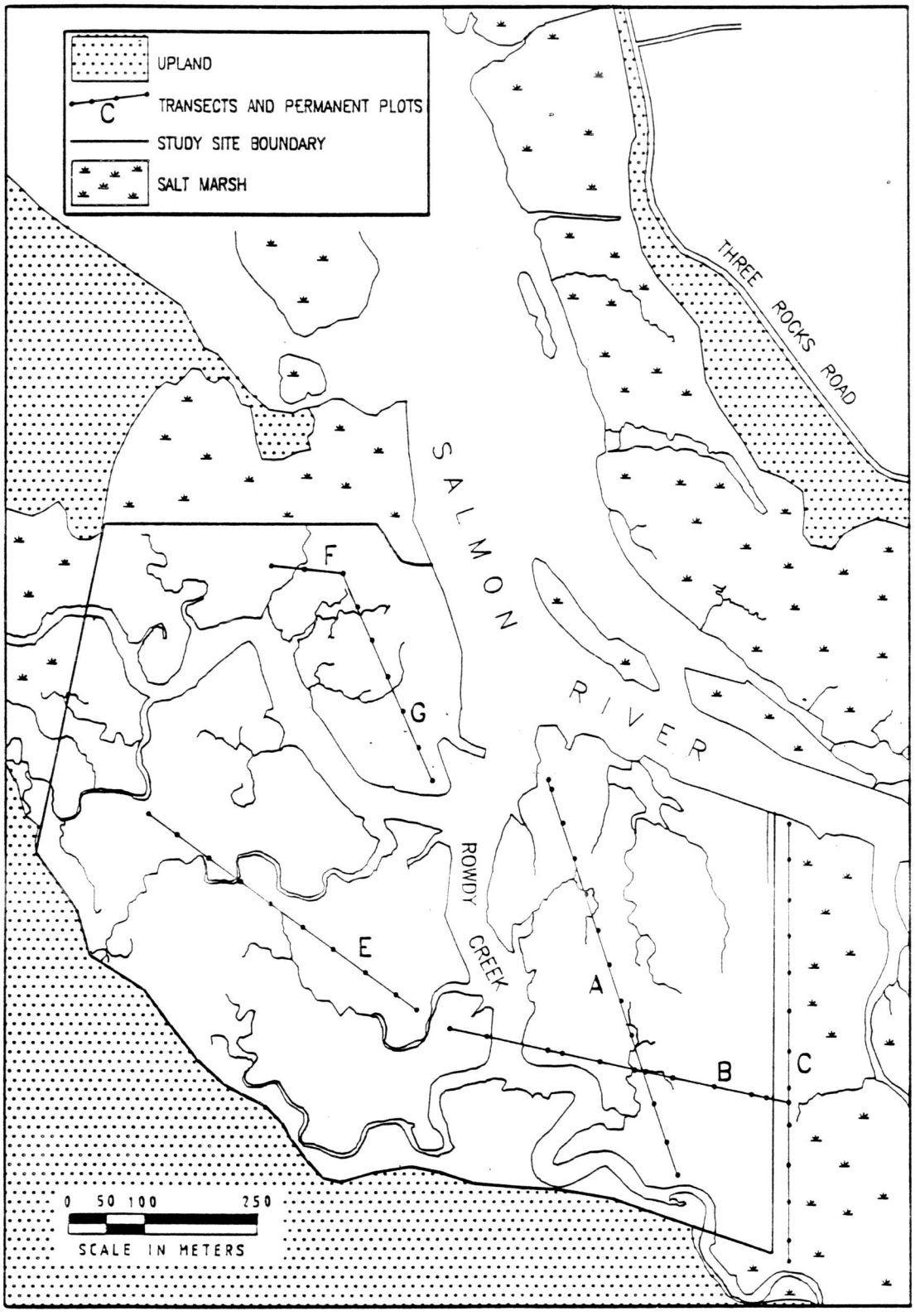


Figure 6. Transects and permanent plots in the Salmon River estuary south shore study site. The central area was returned to tidal connection in August 1987.

In July 1987, about a month prior to dike breaching, we carried out a reconnaissance vegetation survey in the 63 ha Area 57 using 49, 10x10 m^2 plots located so as to typify six vegetation strata based on the vegetation signature on a large scale colored air photo. Species cover class (Table 1) were recorded for each plot.

In Area 57 and 56 (Figure 6), six transects were established in August 1987 along which 66, 1x1 m^2 permanent plots were located; 52 of these plots were in the restored area, 14 were along the Transect C control (Figure 6). Each permanent plot was marked with a 4-6 foot, 1/2 in. PVC pipe. At the base of each permanent plot stake, a 2 foot, 1/2 in. capped PVC pipe was driven to within 2-4 cm of the pasture surface. This capped pipe served as a temporary bench mark for each plot.

On 6 August 1987, all permanent plot elevations along the transects were surveyed by a U.S. Forest Service Hebo District crew. Two elevation measurements were recorded, one at the pasture soil surface, the other on top of the capped PVC stake. Three temporary bench marks were established in the upland.

1988 Field Methods

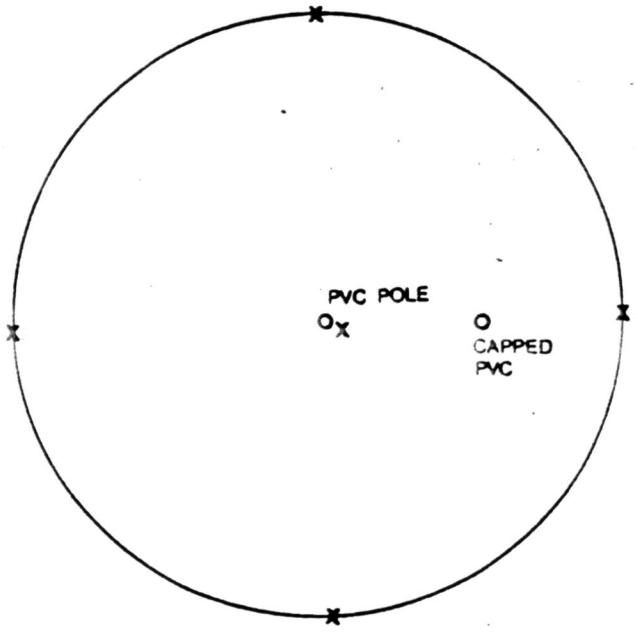
Vegetation was resurveyed in late August at each permanent plot by the same system employed in 1987. This, of course, was the same method used in the north shore study site by Mitchell and continued by us. However, we did not regard 52, 1x1 m^2 permanent plots an adequate sample for the restored area.

The restored area was therefore sampled by a second system aimed specifically at detecting vegetation trend. The **nested frequency method** has been used successfully by range managers in assessing change in species composition (Smith et al. 1986, 1987, Mosley et al. 1987) and is routinely employed by The Nature Conservancy in their monitoring efforts. Advantages of this technique are: (1) sampling is random and therefore can be used as a statistical sample for the entire area; (2) species decisions are objective in that species presence/absence only are recorded (no estimates of cover); (3) sampling is tailored to detect species most sensitive to change; (4) because of the rapidity of sampling, a large number of points can be sampled with little effort.

Three plots sizes were nested, 10x10 cm (0.01 m^2), 50x50 cm (0.25 m^2) and 100x100 cm (1.00 m^2), and species presence/absence recorded in each plot. If a species was present in the smallest plot it was automatically recorded in the two larger-sized plots. To facilitate plot distribution throughout the restored area yet achieve a non-biased sample, nested plots were arrayed by random distance along transects at right angles to the six permanent transects (Figure 6) at the permanent plot position. Four additional randomly oriented 100 m transects with randomly located nested plots were placed in restored areas not sampled by the permanent transects. A total of 688 nested plots were used such that the number of nests were stratified by five cover groups: (1) stressed Potentilla pacifica and/or Atriplex patula with 10-75 % cover (n=268); (2) essentially bare cover with < 10%

cover of P. pacifica and/or A. patula (n=219); (3) vigorous and > 75 % cover of P. pacifica and or A. patula (n=77); (4) Juncus balticus cover > 10 % (n=71); and (5) Cotula coronopifolia cover > 10 % (n=53).

Four permanent bench marks were established in the south shore study site and tied into the north shore site by survey across the river. Surface elevations were remeasured in 1988 at each permanent plot at a minimum of five points centered at the permanent plot stake shown by an "x" in the sketch below. Multiple surface measurements permitted statistical assessment of surface roughness and allowed a more accurate measurement of the restoration area surface than if a single measurement were made.



Surface elevation measurements were also surveyed in control high salt marshes along transects within 15 m of the restoration site in Area 55, north of the restored area, and in Area 56, east of the restored area. These measurements permitted assessment of subsidence and elevation recovery before and after dike removal.

Analytical methods

Plant community data in the restoration area, both prior to and after dike removal, as well as elevation data were analyzed on the south shore using the same multivariate and analytical methods as employed on the north shore study site. Plant community data was based on species cover recorded at permanent plots.

Given that the nested frequency method for trend analysis applied to the south shore restoration area requires at least a second data set to be collected in the future, only raw frequency can be reported here.

RESULTS

Composition of North Shore Restoration Site

Vegetation

In the eleven years since dike removal and tidal water reconnection, vegetation changed dramatically in the restoration site from pasture to salt marsh. Vegetation composition in the adjoining control areas fluctuated, but successional change was not apparent. Compositional changes were traced by changes in major species and by changes in assemblages of species or plant communities.

Species composition changes. Species encountered during the present study together with those recorded by Mitchell (1981) are listed in Appendix 2. Species composition changed during eleven years of restoration. Table 2 shows changes in percent cover and frequency of selected species in the restoration area (Areas 51 and 52 combined) based on approximately 410, 0.1 m^2 plots (20x50 cm plots) located along 12 transects. Figure 7 summarizes changes in major species cover.

In the pre-restoration pasture, two species, *Agrostis alba* and *Potentilla pacifica*, were prominent together with *Holcus lanatus*, *Trifolium repens*, and *Poa trivialis*. The later three and a number of less dominant pasture species were killed within the first year of exposure to saline water. These **upland** species disappeared and are not expected to recur as long as saline conditions prevail. The upland species were replaced by residual and colonizing species.

Table 2. Change in mean percent cover/frequency of selected species in the Salmon River estuary north shore study site from 1978 to 1988; n = number of 20 x 50 cm plots; T = cover <1 %; species acronyms are given in Appendix 2; DM Appendix A = Appendix A in Mitchell (1981).

Year	Bare Grnd.	Species											Source
		AGAL	ATPA	HOBR	CALY	JUBA	SAVI	POPA	DISP	DECE	HOLA	TRRE	POTR	
1978 n=392	T/ 1	37/74	----	T/ 7	T/ 1	5/17	----	41/65	----	----	39/90	9/31	5/57	DM Appendix A
1979 n=408	5/13	29/65	T/ 5	2/13	T/ 5	2/16	T/ 2	29/63	T/ 1	T/ 2	----	----	----	DM Appendix A
1980 n=410	37/60	19/58	4/32	8/13	2/12	2/16	1/11	18/50	T/ 2	T/ 3	----	----	----	DM Appendix A
1981 n=402	16/35	28/66	3/31	4/15	8/24	3/16	6/23	17/49	2/ 5	1/ 4	----	----	----	DM collected data
1982 n=413	7/18	35/73	1/10	3/19	17/41	3/13	12/20	12/44	4/11	3/ 7	----	----	----	DM collected data
1984 n=421	3/ 9	13/57	T/ 3	1/10	47/71	5/20	7/21	12/46	7/17	2/ 9	----	----	----	DM collected data
1988 n=422	----	8/25	T/ 9	T/ 7	63/84	5/19	7/16	6/30	9/21	1/ 6	----	----	----	Our collected data

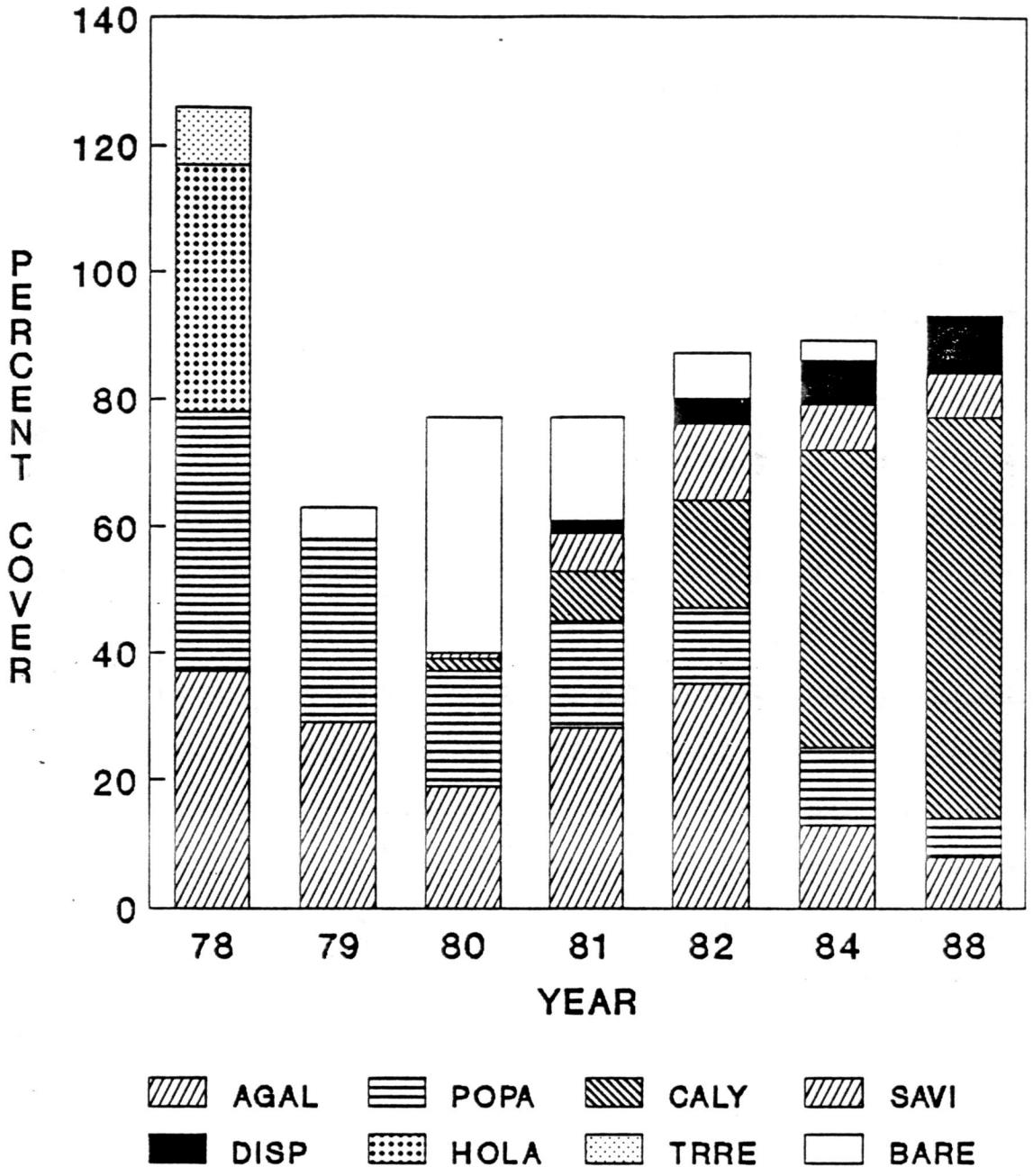


Figure 7. Percent cover of seven important species and percent bare ground in the restoration area of the Salmon River estuary north shore study site from a pasture condition in 1978 to a restored salt marsh in 1988.

Residual species are those that were present in the pasture and persist under saline and/or inundation conditions. A. alba and P. pacifica are the major residuals in the Salmon River restoration. These two have each diminished from about 40 % cover in the pasture to about 8 % cover in the current restored marsh. In 1980-82, A. alba exhibited an increase in cover and frequency prior to appreciable colonization by Carex lyngbyei (Figure 7) suggesting competitive advantage of Carex relative to Agrostis in the restored marsh area.

Colonizers are species that were not present in the pasture but have become permanently established after saline and/or inundation conditions were reimposed. Two classes of colonizers were observed. **Ephemeral colonizers** are those that appeared for a few years and have subsequently died out, such as Cotula coronopifolia and Spergularia marina. These are often annuals and readily increase in bare substrate. **Permanent colonizers** are those species that have become established as continuing components of the restored marsh and are typified by C. lyngbyei and Salicornia virginica. Carex is the overwhelming dominant in the 1988 restored marsh. Table 3 lists selected species based on the classification: upland (pasture), residual, ephemeral, and permanent colonizers.

Exposed bare ground changed during the restoration. Pasture vegetation had no appreciable bare ground. Upon dike breaching in 1979, 5 % of the restored marsh became bare of vegetative cover. Bare ground increased to a maximum of 37 % cover by 1980 and subsequently diminished to no appreciable bare ground in 1988

(Figure 7 and Table 2). Coincident with exposure of bare ground was the colonization by ephemeral species.

Table 3. Percent cover of selected upland, residual, ephemeral, and permanent colonizer species in the north shore restoration site, Salmon River estuary for 1978, 1980, and 1988. Data from 1978 and 1980 are from Mitchell (1981); T = trace.

Species	Upland			Residual			Ephemeral			Perm. colon.		
	78	80	88	78	80	88	78	80	88	78	80	88
Alopecurus geniculatus	7	0	0									
Holcus lanatus	39	0	0									
Poa trivialis	5	0	0									
Ranunculus repens	6	0	0									
Trifolium repens	9	0	0									
Senecio jacobaea	2	0	0									
Agrostis alba				37	19	8						
Hordeum brachyantherum				1	8	1						
Juncus balticus				5	2	5						
Potentilla pacifica				41	18	6						
Cotula coronopifolia							0	T	0			
Puccinelia pumila							0	T	0			
Spergularia marina							0	4	0			
Atriplex patula										0	4	1
Carex lyngbyei										T	2	63
Deschampsia caespitosa										0	T	1
Distichlis spicata										0	T	9
Salicornia virginica										0	1	7

Community changes. Reflecting dramatic turnover in species composition in the restoration area was a corresponding change in plant communities. Two sets of data were analyzed: (1) floristic data in 78, 1 m^2 permanent plots at 20 to 30 m intervals along 19 transects in the restored and control areas; and (2) floristic data from 450, 0.1 m^2 plots at 5 m intervals along the same transects. Because of the advantage of more intense sampling, we report community classification based on the 450 plot-data using two-way indicator species analysis (TWINSPAN).

In the pasture, Mitchell (1981) identified 12 community clusters which she consolidated into three generalized plant communities, Potentilla pacifica, Agrostis alba-Holcus lanatus, and Holcus lanatus. In the surrounding controls, she identified six marsh and transitional communities (Table 4). These relatively intact communities can all still be recognized in the control areas although there have been some shifts in species composition. Presumably, they represent the communities present at the time of pasture diking in 1961-62 and could be construed as the potential marsh development for the restored area.

In 1988, we recognized eight plant communities in the restored and control areas (Area 50, 51, 52, 53) based on TWINSPAN analysis of 450, 0.1 m^2 plots. Mean percent species cover and frequency in these eight communities are summarized in Table 5. These communities are further identified in Table 6 with respect to their elevational position in the marsh and presence in the restored and control areas respectively. Figure 8 maps the distribution of three of these communities and two aggregated community groups based on 450 plots.

Four plant communities dominate the restored area and are minor in extent in the control areas (Table 6): C. lyngbyei community with 100 % membership in the restored area; C. lyngbyei-P. pacifica community with 93 % membership; and A. alba-P. pacifica community with 90 % membership. Three are closely confined to the control areas: Oenanthe sarmentosa-P. pacifica community and Oenanthe sarmentosa-Scirpus microcarpus community.

Table 4. Plant communities, common species, and approximate mean percent cover/frequency in control Areas 50 and 53 in 1978-1979 after Mitchell (1981).

Community	Area	Common Species	Approx. Cov./Freq.
		INTERTIDAL MARSH	
Distichlis-Salicornia	53	Distichlis spicata	42/100
		Salicornia virginica	33/ 95
		Deschampsia caespitosa	12/ 55
		Juncus balticus	8/ 45
		Hordeum brachyantherum	7/ 45
		MIDDLE MARSH	
Juncus-Potentilla-Carex	50,53	Juncus balticus	16/ 96
		Potentilla pacifica	16/ 59
		Carex lyngbyei	10/ 84
		Triglochin maritimum	6/ 88
		Glaux maritima	4/ 71
		Distichlis spicata	4/ 64
Juncus-Deschampsia-Hordeum	50,53	Juncus balticus	41/ 97
		Deschampsia caespitosa	18/ 78
		Hordeum brachyantherum	18/ 81
		Distichlis spicata	10/ 47
		Potentilla pacifica	5/ 33
		HIGH MARSH	
Potentilla-Juncus	50,53	Potentilla pacifica	56/ 93
		Juncus balticus	45/ 98
		Deschampsia caespitosa	5/ 33
		Hordeum brachyantherum	2/ 51
		EXTRATIDAL MARSH	
Potentilla-Aster	50,53	Potentilla pacifica	25/ 68
		Aster subspicatus	22/ 75
		Oenanthe sarmentosa	14/ 30
		Achillea millefolium	12/ 48
		Juncus balticus	10/ 27
		Angelica lucida	7/ 43
		Vicia gigantea	6/ 30
Oenanthe-Scirpus	50,53	Oenanthe sarmentosa	44/ 95
		Scirpus microcarpus	28/ 48
		Potentilla pacifica	14/ 33
		Carex obnupta	5/ 14

Table 5. Mean percent cover/frequency for eight plant communities in the Salmon River estuary north shore study site based on 450, 0.1 m plots in 1988. T = trace (< 1 %); n = number of samples. Species acronyms are given in Appendix 2. Community acronyms are given in Table 6.

Species	Community (n)							
	CALY (171)	CAPO (40)	AGPO (50)	POJU (32)	JUDE (40)	OEPO (12)	OESC (12)	SADI (93)
Caly	**94/100**	**80/100**	18/56	5/16	1/20			9/44
Popa	T/8	**7/75**	**27/94**	**52/100**	**24/78**	**27/83**	1/8	2/10
Agal	T/5	8/53	**27/96**	**9/72**	2/25	1/17		1/7
Juba	T/6	1/33	8/42	**61/88**	**31/95**	**31/75**	1/8	8/34
Dece			T/4		**30/80**	T/8		4/25
Oesa				3/3		**39/100**	**47/100**	
Scmi					T/3		**49/75**	
Disp	1/4	T/5	T/4		15/50			**37/94**
Savi	2/3				1/3			**37/86**
Atpa	T/2	1/18	1/14	T/9	1/38	T/25	1/17	1/25
Hobr	T/1	T/5	2/30	T/9	1/43			1/11
Trma	T/2		1/4	1/6	3/20			4/16
Lapa				2/28	T/10	2/50	T/8	
Gaap		T/3	T/4	T/13				
Gatr			T/2	T/9	T/3	T/8		
Assu			2/8	8/38	2/10	T/17		
Acmi			T/2	6/22	1/8			
Hola			T/2	T/6				
Trwo			T/6					1/2
Caob				T/3		30/50	T/8	
Feru				T/6	T/8	T/8		
Anlu				2/16	T/5			
Vigi				1/3				
Fear						T/17	10/25	
Visa						T/17		
Grin								T/8
Cusa								T/8

Table 6. Eight plant communities and community acronyms in the north shore Salmon River study site in 1988 with respect to generalized tidal position and percent membership in the restored and control areas.

Community	Community Acronym	Tidal Position	Percent Membership	
			Rstrd Area	Cntrl Area
Carex lyngbyei	CALY	Intertidal	100	0
Salicornia virginica-Distichlis spicata	SADI	Intertidal	76	24
Carex lyngbyei-Potentilla pacifica	CAPO	Middle	93	7
Agrostis alba-Potentilla pacifica	AGPO	Middle	90	10
Potentilla pacifica-Juncus balticus	POJU	High	41	59
Juncus balticus-Deschampsia caespitosa	JUDE	High	13	87
Oenanthe sarmentosa-Potentilla pacifica	OEPO	Extratidal	0	100
Oenanthe sarmentosa-Scirpus microcarpus	OESC	Extratidal	0	100

Salt marsh vegetation commonly has strong zonal expression reflecting gradients in elevation (tidal inundation) and salinity. This zonal pattern was evident in both control areas flanking the restoration area in 1978 and could still be seen in the control areas in 1988 as shown in the map of plant communities (Figure 8). Zonation is evident in the restored area where the Potentilla and Agrostis communities are starting to characterize the upper elevation edges of the marsh. Both these species were residuals but the communities are presently being reconstituted as result of occasional tidal flooding. On the other hand, the Juncus and Deschampsia communities, key features in this zonal pattern of communities which are characteristic in the control areas, have no counterpart in the restored area as yet.

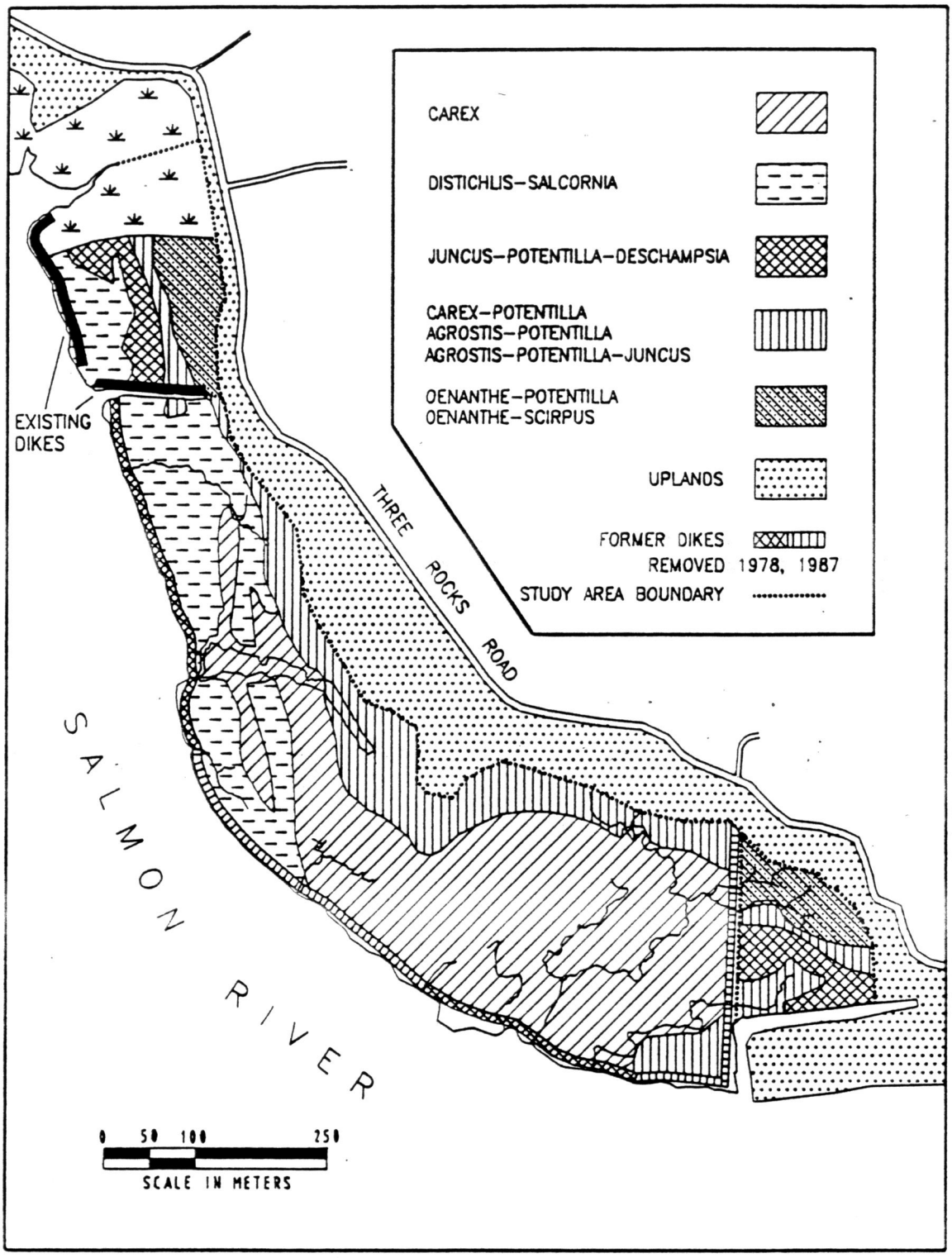


Figure 8. Distribution of plant communities in 1988 in the restored and control areas in the north shore study site of the Salmon River estuary.

Structure of the North Shore Study Site

Elevation as a Structural Control

Wetland character and development are controlled by hydrology and water table position. In coastal salt marshes, elevation determines this position by controlling the frequency and period of tidal inundation on a daily and seasonal basis and therefore elevation is a primary control on most marsh processes. Study site surface elevations were surveyed by Mitchell in 1978 and 1979. We resurveyed many of the same locations in 1988. Mitchell registered all elevations to the Mean Lower Low Water (MLLW) tidal datum and established a number of bench marks (Appendix 1) which we recovered in 1988. In the present study, we registered all elevations to meters above National Geodetic Vertical Datum (NGVD) based on a USGS 1987 first order survey along the Pacific Coast. For comparison purposes, Mitchell's elevations are also reported here relative to the NGVD.

In this section we discuss (1) pre-dike (natural marsh) elevations; (2) pasture subsidence upon diking; (3) 1988 elevation survey; (4) response of surface elevation to dike removal and subsequent reconnection to tidal circulation; (5) elevation distributions of plant communities and species; (6) surface roughness; and (7) vegetation established on former dikes.

Pre-Dike Elevation and Subsidence

Historical elevations. Study site elevations prior to dike construction in 1962 were unknown; however, local natural marsh elevations may be inferred from those surveyed by Mitchell in

1978-1979 in control Areas 50 and 53 and from her measurements on the south side of the Salmon River in non-diked marshes. Mitchell made two assumptions: (1) that the pre-dike study site marsh was at the same elevation as the nearby transitional and extratidal (high) salt marshes and (2) that changes in non-diked natural marsh elevations were negligible during the 17 years of dike history. Based on these assumptions, Mitchell (1981) mapped elevation relationships in the diked (i.e., breach or restored) portion of the study site and the adjacent non-diked intact marshes (controls) as shown in Figure 9. The diked area ranged from 0.90 m MLLW (0.95 m NGVD) to 1.50 m MLLW (1.55 m NGVD), mostly corresponding to intertidal marsh (Table 7). Intact nearby salt marshes in the same relative position of the diked marsh, which Mitchell judged as transitional or extratidal, ranged from 1.30 m MLLW (1.35 m NGVD) to 1.75 m MLLW (1.80 m NGVD). In other words, the diked marsh subsided appreciably during 15 years of dike history.

Subsidence. Mitchell observed that over the 15 years of dike history, diked pastures subsided 30 to 40 cm relative to the "control" marshes. She attributed this in part to (1) grazing and mowing causing compaction leading to increased soil bulk density in the diked area; (2) draining together with reduced tidal inundation resulting in oxidation of organic matter in the upper strata of soil; and (3) loss of the buoyant effect of water after diking. She detected subsidence with three kinds of evidence. First, she detected subsidence from paired surface elevation measurements of the diked and "control" side of the flanking dikes on both sides of the river. Second, she employed

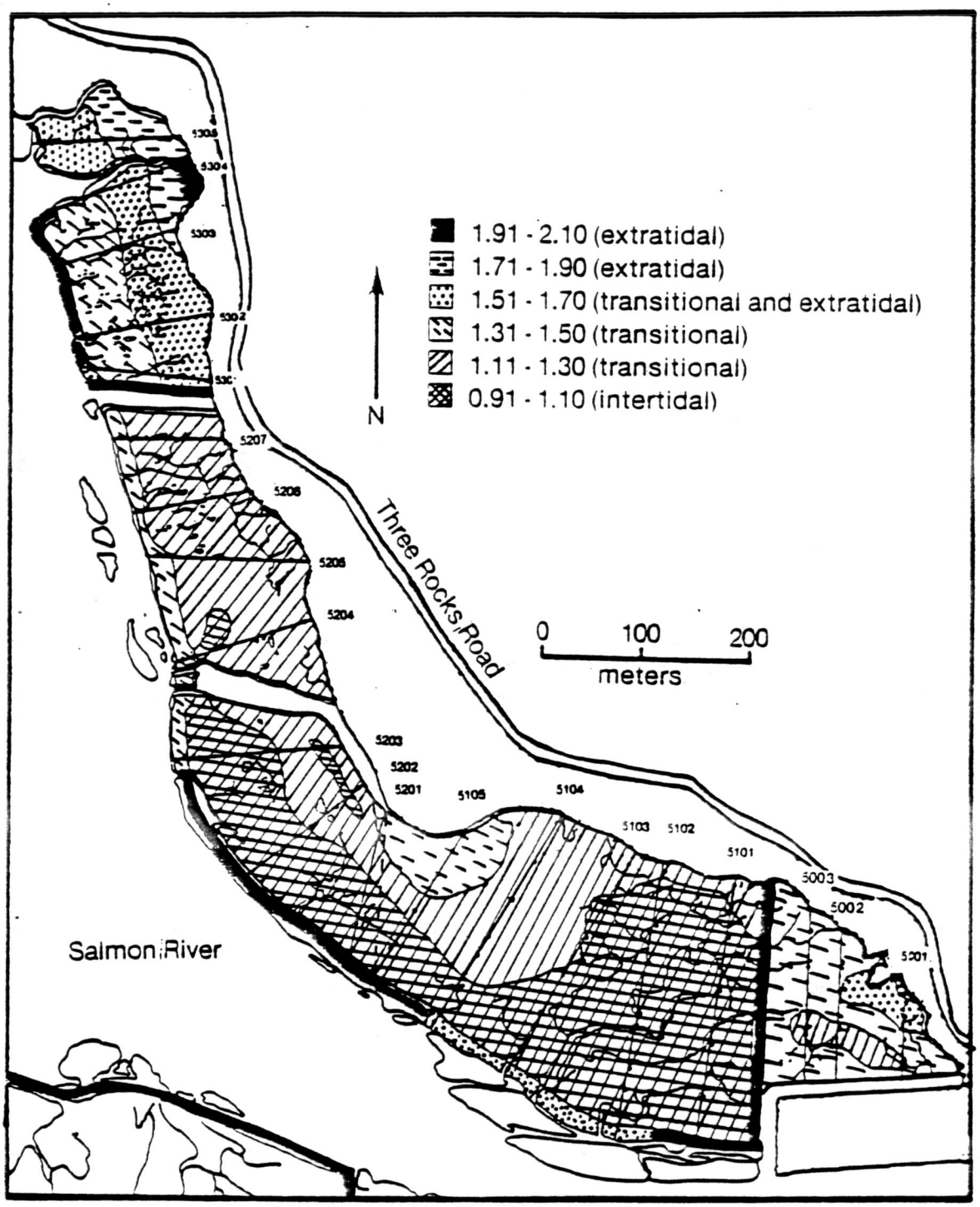


Figure 9. Elevation zones in the Salmon River estuary north shore study site in 1979 (from Mitchell 1981:116).

Table 7. Elevation zones surveyed in 1978 and 1979 in the Salmon River study sites (after Mitchell 1981).

Elevation Range		Elevation Zone
NGVD (m)	MLLW (m)	
1.96 - 2.15	1.91 - 2.10	extratidal
1.76 - 1.95	1.71 - 1.90	extratidal
1.56 - 1.75	1.51 - 1.70	high transitional & extratidal
1.36 - 1.55	1.31 - 1.50	mid transitional
1.16 - 1.35	1.11 - 1.30	low transitional
0.96 - 1.15	0.91 - 1.10	intertidal

elevation measurements of a cross-section of Creek 3 on the diked and "control" sides of the flanking dike between Area 50 and 51. Third, she used measurements of soil thickness above a layer of organic matter on either side of the dike.

In 1988 we reassessed Mitchell's analysis. Mean elevation of intact (i.e., control) salt marsh differed between the north shore study site and south shore study site based on all measurements (Figure 10). Mean elevations of the control areas on the south shore study site was 1.32 m NGVD (0.055 sd); on the north shore controls it was 1.53 m NGVD (0.143 sd). This 20 cm difference is partly accounted for by measurements on the north side taken along profiles extending across transitional and extratidal marsh communities, while on the south side, control measurements were narrowly confined to transitional marsh.

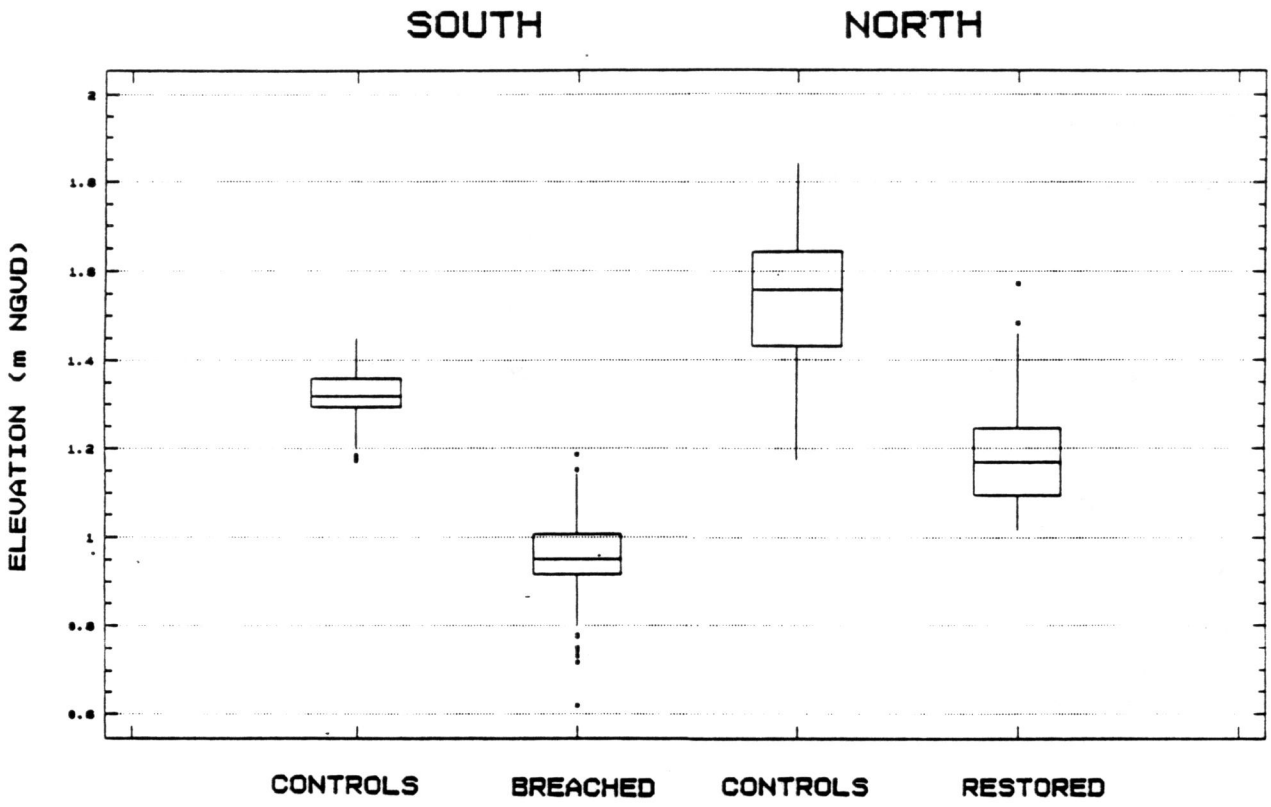


Figure 10. Elevation differences between diked pasture (breached and restored) and undiked salt marsh (controls) on the south (n=218) and north (n=196) shore study areas, Salmon River estuary in 1988.

Adjusting for the same marsh type (transitional marsh) on the north and south side, the north shore study site control mean elevation was 1.46 m NGVD (0.074 sd) compared with the south shore control elevation of 1.32 m NGVD (0.055 sd), indicating that transitional marsh types on the north side were 14 cm higher than on the south side. It is difficult to ascribe these elevation differences to a particular process, poor control on sampling, or to some other factor.

Although there were elevation differences between the north and south side of the river, **relative subsidence** on both the north and south side was approximately the same (Figure 10), 34.87 and 36.69 cm respectively. The difference in subsidence between the north and south sides is partly attributable to the fact that these measurements were in 1988. By 1988, the north side restored area was 10 years old. On the south side, the restored area was one year old; therefore, the older restoration area has regained some of its former elevation, a consideration discussed further below.

Elevation Surveys

1978-79 Surveys. Mitchell surveyed elevations at 5 m intervals along 20 transects in the north shore control Areas 50 and 53 and restoration Areas 51 and 52. The restoration areas were surveyed in 1978 before dike breaching and again after breaching in 1979. Control Area 50 was surveyed in 1978 and control Area 53 in 1979. Measurements along transects included permanent plot elevations. She produced an elevation zone map from this data exhibiting six zones corresponding to generalized tidal marsh types (Figure 9). An extensive area in the up-stream section of the restoration site was intertidal, particularly in the vicinity of the major drainage creek. The down-stream section of the restoration site was low transitional marsh. Control Areas 50 and 53 were largely upper transitional and extratidal. Recently leveled dikes were in the mid to upper transitional elevation range. Mitchell characterized pre-dike

plant communities and newly established salt marsh communities by elevation zones and predicted that "... because of 30 to 40 cm subsidence of the study site ... during the 15-year period that it was diked, somewhat lower marsh communities than the pre-diking transitional and extratidal communities are expected to develop after dike breaching. Intertidal and low transitional marshes should be the most common types" (Mitchell 1981: 115).

1988 Survey. We conducted a detailed and carefully controlled survey in 1988, the results of which are reported here. Figure 11 shows north shore study site topography at 5 cm elevation intervals in 1988. A low area can be seen in the southernmost section of the restoration area and generally along the river side of the marsh, as would be expected. A low area also occurs in northern control Area 53 at a point where a small creek forms a connection to the estuary at a breach in the old abandoned dike fronting this control. The highest portions of the study area are near the upland boundaries of the two controls, both of which receive freshwater runoff. Another high occurs at the base of the central lobe of upland. The slightly higher elevations of the two controls relative to the restoration area is also seen by sharp elevation gradients bounding the restoration area.

Elevation changes 1978-88. The study site elevation pattern described by Mitchell for 1978-79 (Figure 9) corresponds with

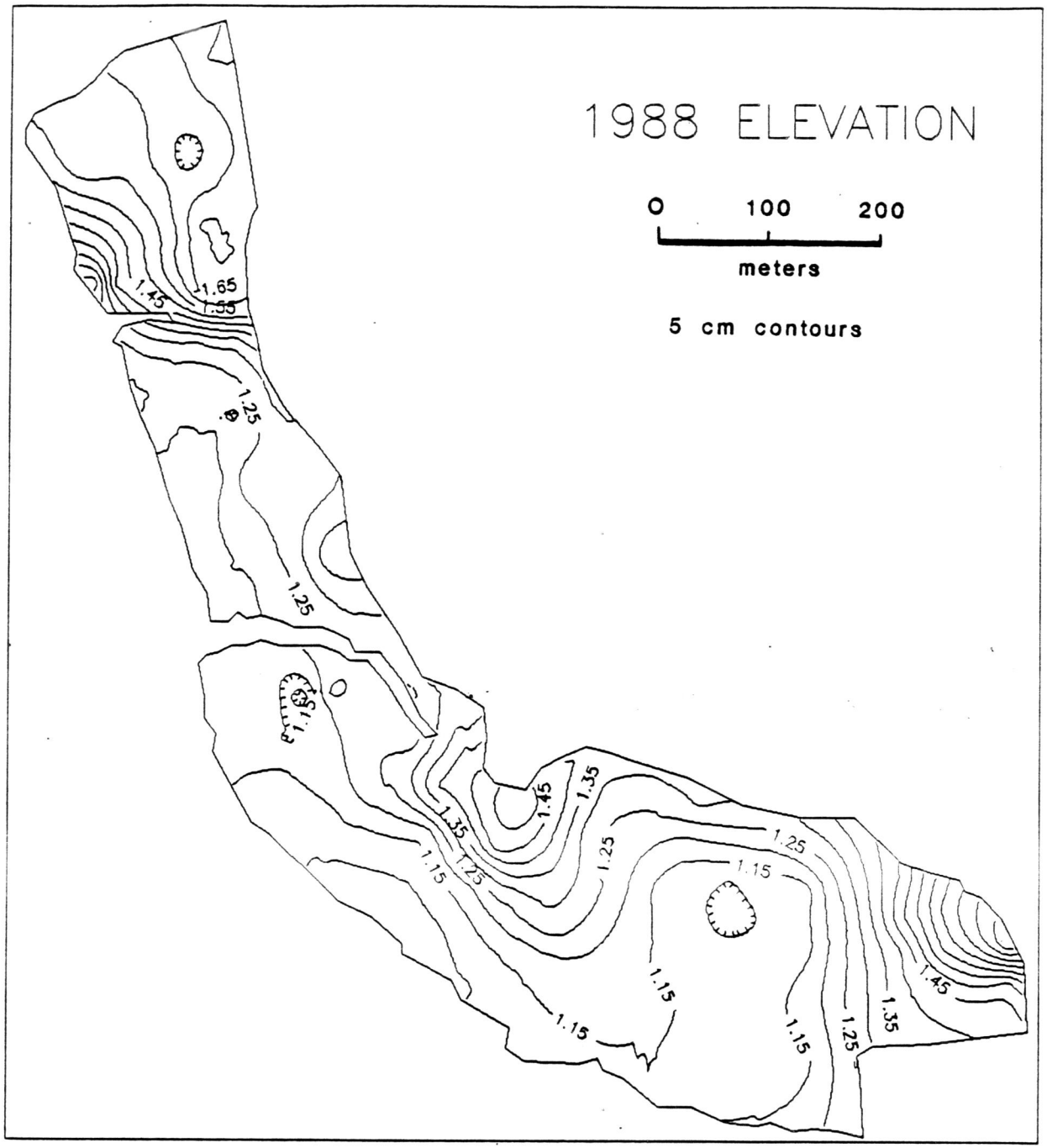


Figure 11. Topography of the north shore study site, Salmon River estuary in 1988; contour interval 5.0 cm.

that mapped in 1988 (Figure 11). We surveyed and mapped elevation changes at permanent plot locations (Figure 12). Almost all locations increased in elevation over the 10 year restoration period. The northern control Area 53 surface elevation increased less than 2 cm, while the upper portions of the southern control Area 50 increased 6 cm or more. We account for these differences by the effect of strong freshwater input in control Area 50 together with moderate tidal exchange (bringing in sediment and facilitating organic accumulation) compared with control Area 53 which had restricted tidal exchange and slight freshwater seepage.

Figure 13 shows selected elevation profiles along transects in the restored area. Elevations that we surveyed in 1988 tracked those in 1978 and 1979. The western part of the restored area, dominated by Creeks 1, 2, and 3 and covered heavily by C. lyngbyei, showed increases of 4 to 7 cm (Figure 13 a,b). In the center portion of the restored area, elevation increases diminished to less than 2 cm (Figure 13 c,d). In the northern portion of the area changes again increased to 3 to 5 cm (Figure 13 e,f). The strongest increases occurred where tidal exchange was greatest, namely near creek systems. One possibility explaining this relationship is that tidal exchange leads to siltation. Another, probably more important, is that buoyancy and thus elevation increased where there was greater saturation.

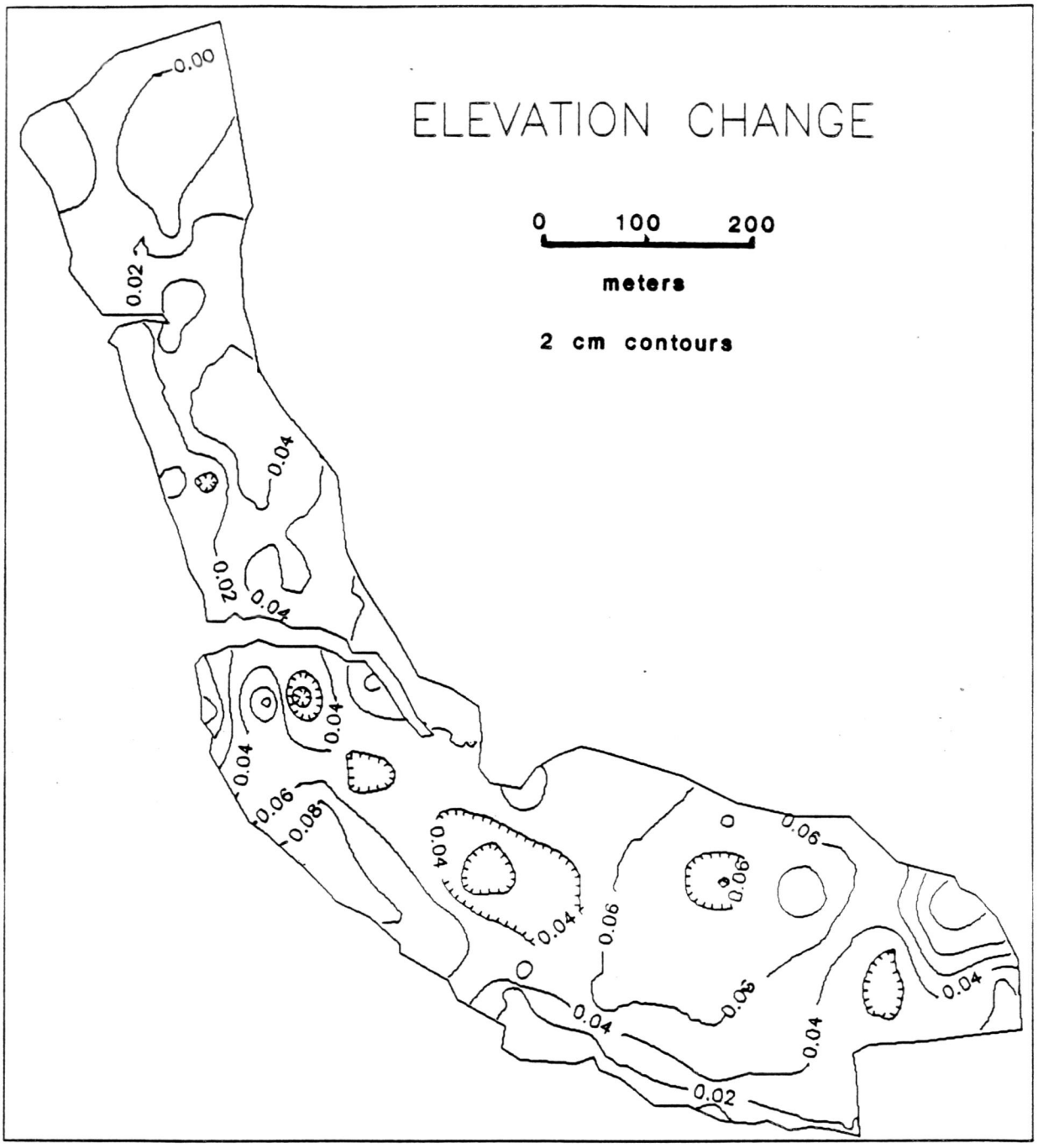


Figure 12. Elevation change from 1978 to 1988 in the north shore study site, Salmon River estuary.

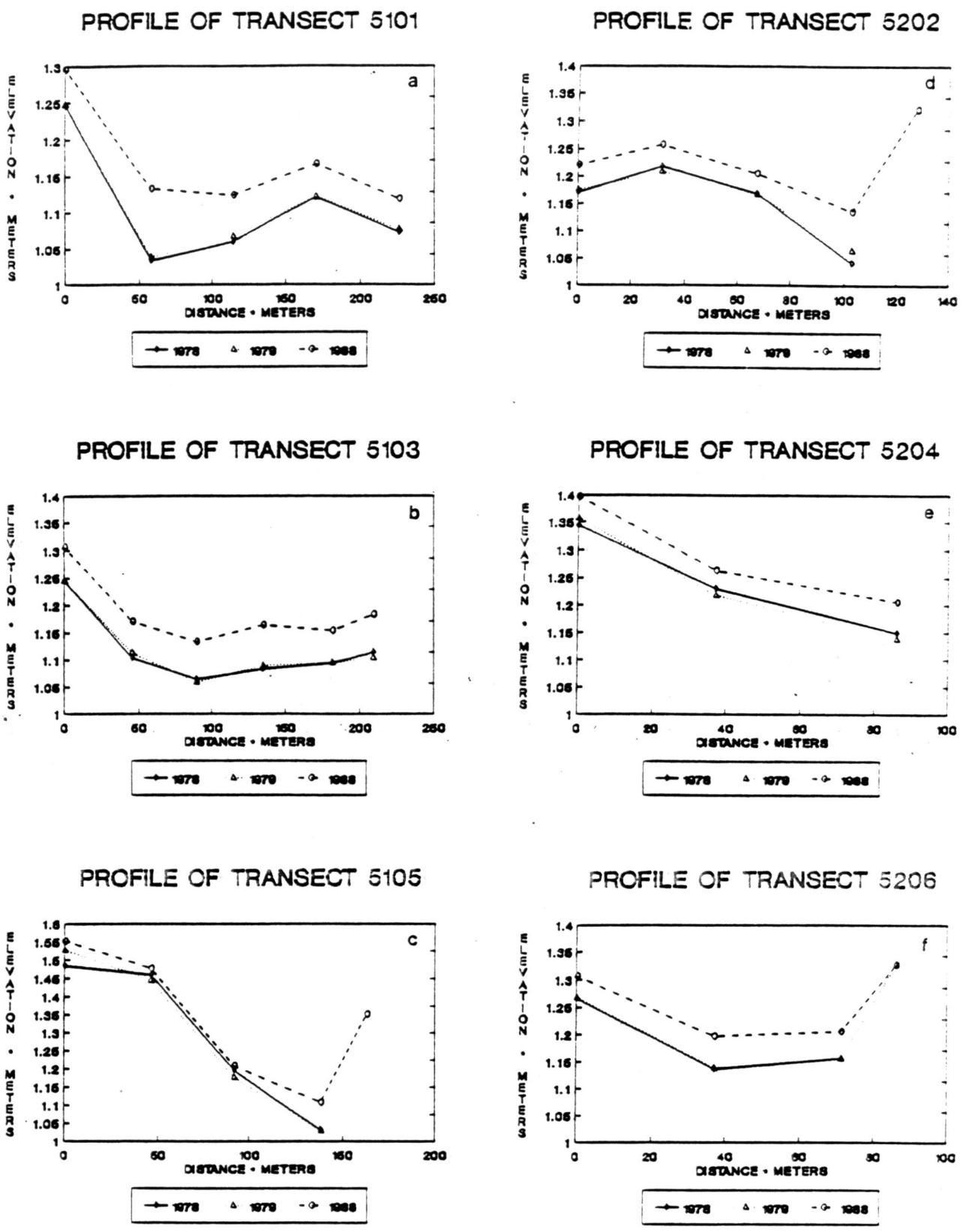


Figure 13. Selected elevation profiles along transects in restored area of the north shore study site, Salmon River estuary.

Permanent plots on dikes breached in 1978 were treated as a separate data set. Elevation change with dike plot elevation exhibited the same relationship as observed in the restored area; namely, the lowest plots on the dike increased more in elevation than did the highest dike plots.

Elevation changes differed between the two control areas (Area 50 and 53). The upper portion of Area 50 increased 6 to 10 cm in elevation while the lower parts of the area increased 1 to 2 cm. In the control Area 53, elevations increases were negligible, or in some locations negative. As mentioned earlier, Area 53 had restricted tidal exchange and little freshwater input. Better tidal exchange and strong freshwater input marked Area 50 and may account for these differences.

The relation between plot elevation and change in plot elevation over the 10 year restoration period (1978-1988) is shown in Figure 14. In the restored area, changes were greatest (8 to 10 cm) at the lowest elevations and diminished as elevation increased ($r=0.62$, se 0.017). In the control areas, there was a much greater scatter in values and we are hesitant to generalize about the relationship. A number of plots in control Area 53 exhibited a decrease in elevation over ten years, possibly due to erosion or excavation by rafted drift logs.

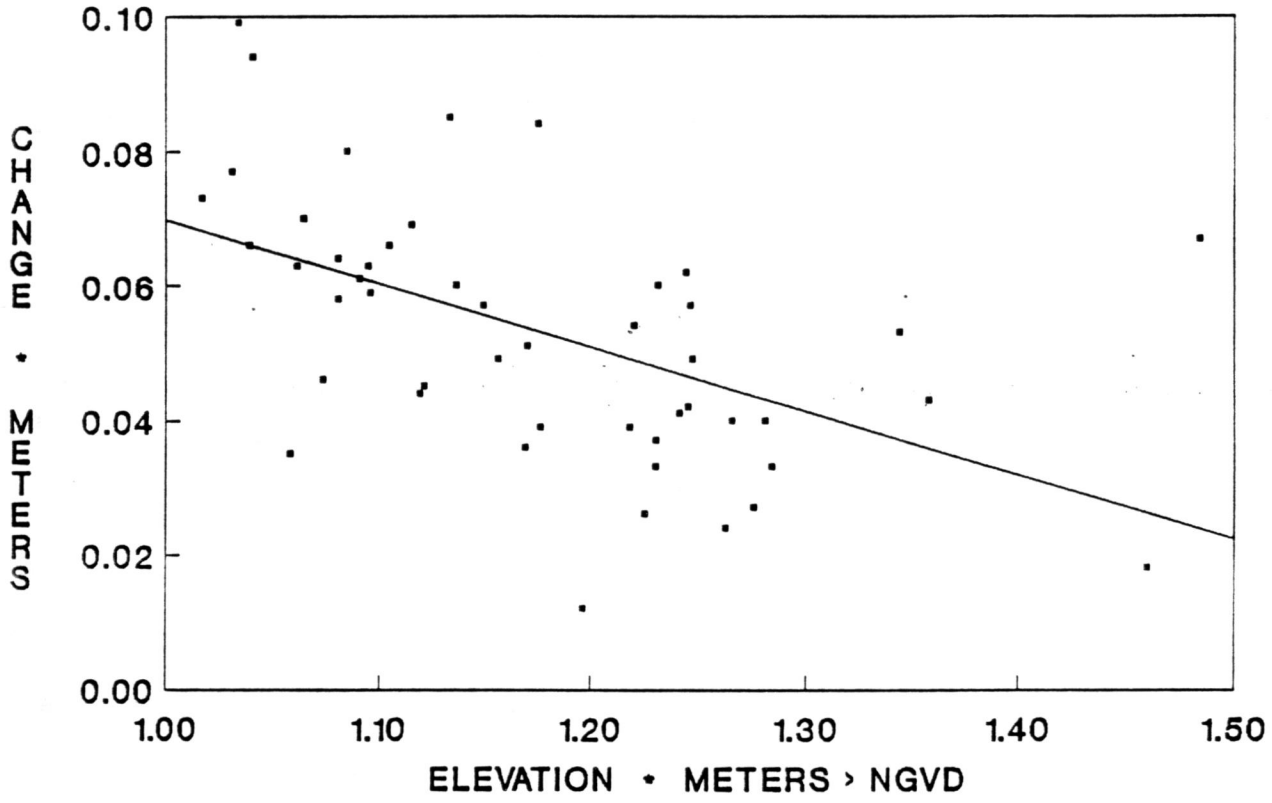


Figure 14. Relation between elevation of permanent plots and change in elevation of permanent plots from 1978 to 1988 for the restored area in the north shore study site, Salmon River estuary; r=0.62, se=0.017.

Elevation Vegetation Relationships

Communities. We identified six plant communities in the north shore study site in 1988; two were widespread in the restoration area (*Carex lyngbyei* and *Salicornia virginica* communities) and three occurred in the control areas or at the upper margin of the restoration area. In addition, we recognized four communities on dikes leveled in 1978, which for purposes of elevation analysis were grouped as a single "dike community".

Plant communities differed in elevation (Figure 15 and Table 8 and 9). For comparison we measured four elevations at each permanent plot. Both the *Carex* and *Salicornia* communities were intertidal and poorly represented in the control salt marshes flanking the restoration area. The wet and dry phases of the *Potentilla pacifica* community differed slightly in elevation and we regarded these as transitional communities between upland and salt marsh. Both were well represented in the control areas. We considered the *Oenanthe sarmentosa* community a freshwater wetland at the upper margin of the two control areas, a common occurrence in salt marshes throughout the Pacific Northwest (Frenkel et. al. 1978). Mean elevations of the communities were all significantly different ($p=.001$) except the "dike community" and *Potentilla communities* (Table 8) which were close in their elevation distribution but differed in substrate characteristics.

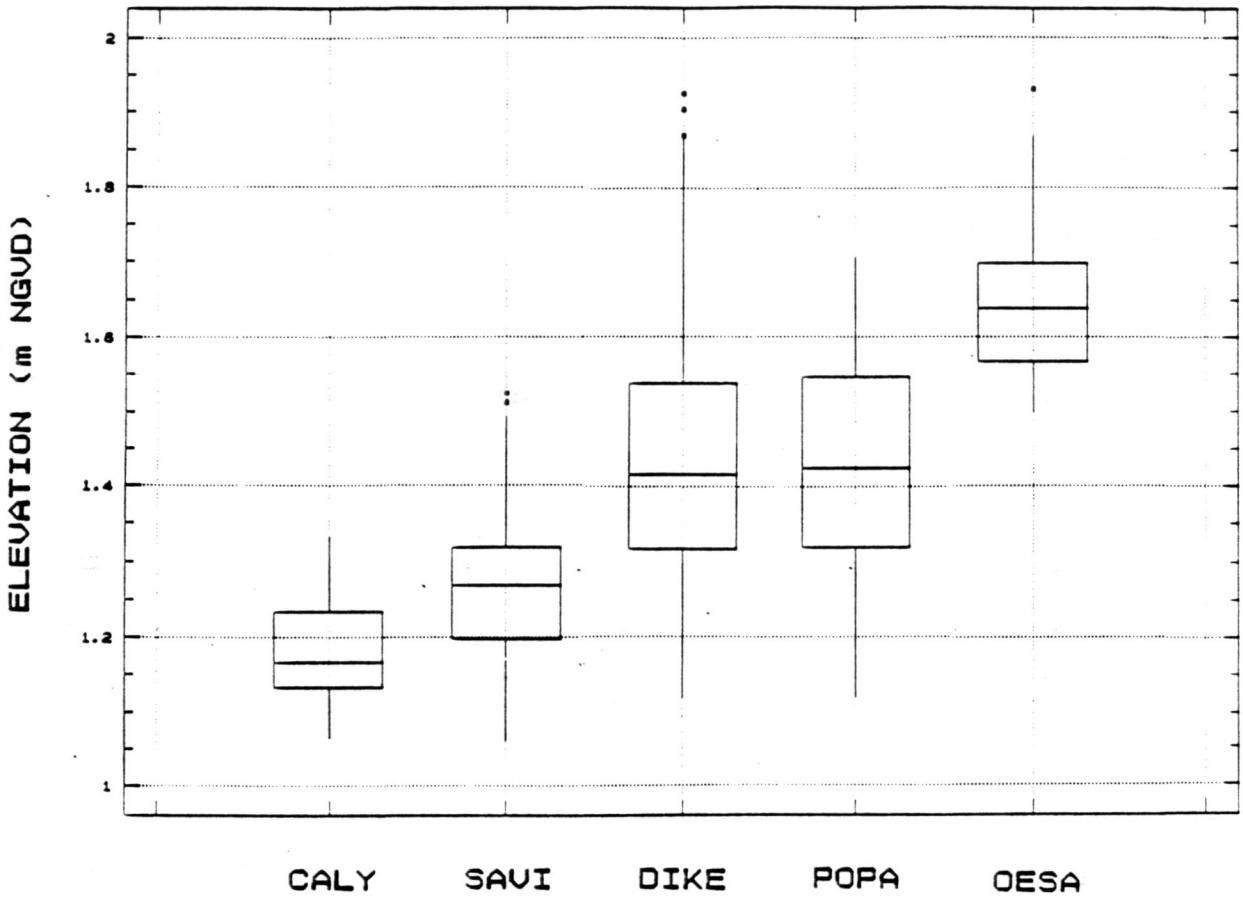


Figure 15. Elevation distribution of four plant communities and leveled dikes in the restoration area of the north shore study site, Salmon River estuary in 1988; CALY = Carex lyngbyei comm., SAVI = Salicornia virginica-Distichlis spicata comm., POPA = wet and dry phases of the Potentilla pacifica comms., OESA = Oenanthe sarmentosa comms; details in Table 8.

Table 8. Comparison of plant community elevations in m NGVD in the Salmon River North Shore Study Site in 1988.

Community	n	Mean Elev.	Std. Error	Std. Dev.	Range
Carex lyngbyei	120	1.184	0.0006	0.067	1.065-1.333
Salicornia virginica	60	1.207	0.0135	0.104	1.062-1.525
Potentilla pacifica WET	36	1.359	0.0224	0.134	1.119-1.596
Potentilla pacifica DRY	60	1.472	0.0163	0.126	1.242-1.707
Oenanthe sarmentosa	31	1.651	0.0195	0.109	1.498-1.931
Dike Communities	84	1.429	0.0199	0.182	1.118-1.927

Table 9. Matrix of two-tailed t-test values for plant community elevation means in the Salmon River Study Site in 1988. The wet and dry phases of the Potentilla community were treated as a single community. Asterisk indicates significance at p=0.001. Mean elevation in m NGVD.

PLANT COMMUNITY		CALY	SAVI	POPA	OESA	DIKE
MEAN ELEV	COMM					
1.184	CALY	----	*6.378	*16.935	*29.969	*13.464
1.207	SAVI		----	*7.780	*16.414	*6.213
1.359	POPA			----	*8.047	0.029
1.651	OESA				----	*6.363
1.429	DIKE					----

Species. Species exhibited broader elevation ranges than did communities. Mean elevations of the 12 most important species based on their occurrence throughout the restoration area are given in Table 10. Elevation distributions of eight of these species are depicted in Figure 16. Generally, these distributions approximate normal distributions and t-tests are therefore a valid way for comparing means. Table 11 displays two-tailed t-test values of means and indicates that the dominant species form two elevation groups: an intertidal group with Carex lyngbyei, Salicornia virginica, Triglochin maritimum and Distichlis spicata; and a transitional group consisting of Potentilla pacifica, Juncus balticus, Agrostis alba, and Deschampsia caespitosa. With the exception of Distichlis, individual species within these groups shared the same elevation ranges.

Table 10. Mean elevation of important species in the restoration area of the north shore study site, Salmon River estuary in 1988.

Species	n	Mean Elev. (m NGVD)	Median Elev. (m NGVD)	std. error	std. dev.
Carex lyngbyei	72	1.270	1.265	0.015	0.124
Salicornia virginica	22	1.306	1.305	0.021	0.097
Triglochin maritimum	23	1.319	1.313	0.019	0.089
Distichlis spicata	36	1.325	1.310	0.021	0.127
Potentilla pacifica	61	1.404	1.372	0.023	0.179
Juncus balticus	59	1.409	1.372	0.020	0.150
Deschampsia cespitosa	42	1.411	1.414	0.023	0.150
Agrostis alba	51	1.415	1.397	0.026	0.188
Festuca rubra	12	1.454	1.392	0.049	0.169
Aster subspicatus	15	1.625	1.635	0.036	0.140
Oenanthe sarmentosa	8	1.628	1.620	0.041	0.116
Holcus lanatus	9	1.666	1.602	0.066	0.198

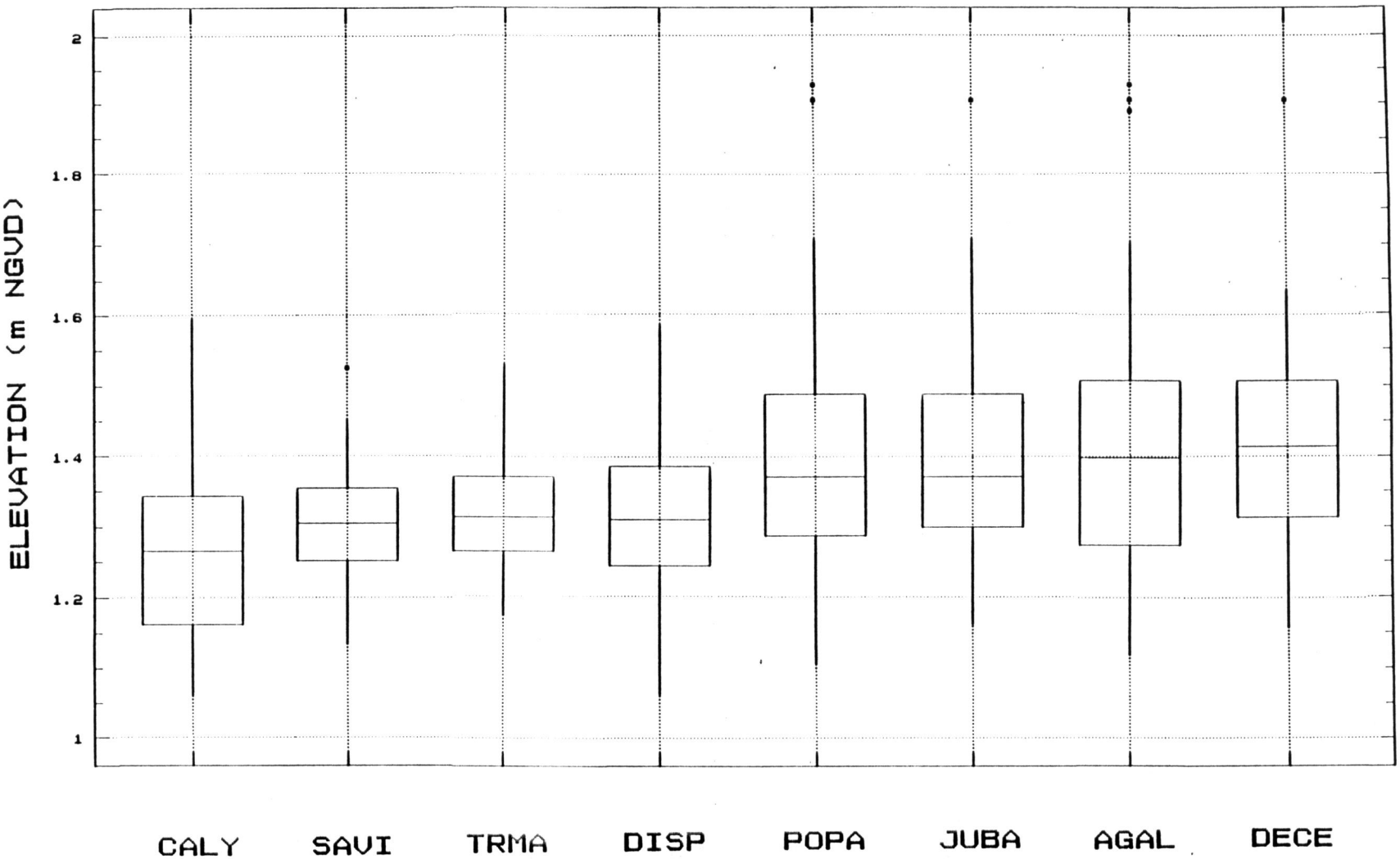


Figure 16. Elevation distribution of eight important species in the restoration area of the north shore study site, Salmon River estuary; CALY = Carex lyngbyei, SAVI = Salicornia virginica, TRMA = Triglochin maritimum, DISP = Distichlis spicata, POPA = Potentilla pacifica, JUBA = Juncus balticus, AGAL = Agrostis alba, DECE= Deschampsia cespitosa.

Table 11. Matrix of two-tailed t-test values for important salt marsh species and their elevation means in the Salmon River Study Site in 1988. Asterisk indicates significance at p=0.001. Mean elevation in m NGVD. Species acronyms are give in Appendix 2.

Mean Elev.	Spp.	CALY	SAVI	TRMA	DISP	POPA	JUBA	DECE	AGAL
1.270	CALY	----	1.23	1.73	**2.16	**5.06	**5.80	**5.40	**5.15
1.306	SAVI		----	0.47	0.63	**2.44	**3.00	**2.98	**2.58
1.319	TRMA			----	0.22	**2.18	**2.71	**2.70	**2.33
1.325	DISP				----	**2.31	**2.79	**2.69	**2.48
1.404	POPA					----	0.17	0.20	0.31
1.409	JUBA						----	0.06	0.18
1.411	DECE							----	0.11
1.415	AGAL								----

Surface Roughness

Marsh surface roughness was evident and differed from area to area in the study site (Table 12). Generally, the lowest areas dominated by Carex were smoothest (variance 0.003 m); higher areas with Agrostis and Potentilla, which also were disturbed more by rafted material, were rougher (variance 0.016 m); control areas as well as dikes leveled in 1978, with tussocks of Deschampsia and stranded debris, were quite rough (variance 0.039 m).

Table 12. Variation in surface roughness in the Salmon River north shore study site in 1988 based on four elevation measurements per plot in meters.

Area	n	Mean Std. Dev.	Mean Variance
Restoration Area 51	28	0.046	0.003
Restoration Area 52	24	0.090	0.011
Combined Area 51 and 52	52	0.066	0.007
Control Area 50	11	0.087	0.018
Control Area 53	16	0.100	0.015
Dikes leveled in 1978	20	0.156	0.039

Leveled Dike Vegetation and Elevation

Leveled dike vegetation constituted a relatively small portion of the Salmon River north shore study site (Figure 5). Most of the site was either restored marsh or control marsh. However, since the leveled dikes provide a scraped surface with elevations comparable to those found in high marshes, vegetation developed on these former dikes may provide useful information for the individuals who might be designing restoration projects. Two types of leveled dike vegetation were present: (1) plant cover on freshly leveled dikes (leveled in 1987) and (2) plant cover on old dikes leveled in 1978.

Recently leveled dikes (1978-1980). In 1980, Mitchell (1981) identified four vegetation types on dikes two years after dike leveling: (1) a **Spergularia marina Type** with the low marsh colonizers Salicornia, Carex, Agrostis, Cotula coronopifolia, and Puccinellia pumila (at elevations 1.10 to 1.15 m NGVD); (2) a **Deschampsia Type** with Distichlis occupying somewhat higher ground (1.35-1.45 m NGVD); (3) an **Upland Species Type** with Festuca arundinacea, Agrostis tenuis, Lotus corniculatus, and Holcus lanatus on the highest portions of the leveled dikes (1.55-1.70 m NGVD); and (4) a **Bare Ground Type** found on the driest and most compacted soils at middle elevations (1.45-1.55 m NGVD). All of these types were comprised of species that colonized the freshly leveled surfaces; however, whether the propagules were residual, blew in, or were washed in, is not known. In 1988, vegetation assemblages colonizing dikes leveled in 1987 were very similar to the types described by Mitchell and therefore these very recently colonized dikes are not further discussed.

Old leveled dikes (1978-1988). Vegetation on old dikes leveled in 1978, however, was evaluated and is discussed here. In 1988, eleven years after dike breaching, many of the same vegetation types recognized by Mitchell in 1980 could still be identified and elevation patterns were similar. Using vegetation data collected from 1 m^2 permanent plots, we classified four plant communities aggregated into three dike vegetation types (Table 13). (1) The **Successional Salt Marsh Type** occupies elevations between 1.10 and 1.50 m NGVD and comprises a Carex-Distichlis community and Carex-Deschampsia community. Analysis with 0.1 m^2 Daubenmire plots showed that a Salicornia-Distichlis community was also present in the more saline and more sandy areas. (2) The **Mature High Salt Marsh Type** occupies elevations from 1.30 to 1.55 m NGVD and was represented by a Juncus-Deschampsia community comparable in elevation and composition to that of the high salt marsh in the control areas. (3) The **Transitional Salt Marsh Type** occupies the highest elevations (1.55-1.70 m NGVD) and is comprised of many upland species mixed with some salt marsh species.

The Mature High and Transitional Salt Marsh Types were at elevations that were infrequently flooded by salt water. It appears that given a surface elevation in the range of 1.40 to 1.70 m NGVD with the possibility of infrequent tidal flooding, and a local sources of propagules, natural **high** salt marsh vegetation can develop after about 10 years. In the study area, this development only occurred at the upper fringes of the restoration area **and** on leveled dikes.

Table 13. Dike vegetation types and plant communities on dikes leveled in 1978 based on 1988 floristic data in the Salmon River estuary north shore study site. Numbers in table refer to cover classes: 5 = >75% cover, 4 = 50-74% cover, 3 = 25-49% cover, 2 = 2-24% cover, 1 = <2% cover, - = absent.

Dike Vegetation Type	**SSM**		**MHSM**	**TSM**
Plot Number	222 637 455	222111 344234 645677	2122 5357 6854	111 243 789
Salicornia virginica	4--	2-----	----	---
Distichlis spicata	544	--3---	2-33	---
Triglochin maritimum	-3-	------	1---	---
Carex lyngbyei	244	354545	3434	---
Grindelia integrifolia	---	------	---1	---
Spergularia macrotheca	---	------	1---	---
Deschampsia caespitosa	-12	444---	3413	11-
Agrostis alba	-1-	1131-1	----	111
Hordeum brachyantherum	---	------	-211	1-1
Juncus balticus	---	2--1-1	4441	444
Potentilla pacifica	---	---2--	-331	441
Atriplex patula	11-	------	----	11-
Angelica lucida	---	------	----	244
Aster subspicatus	---	------	----	132
Festuca arundinacea	---	------	----	123
Festuca rubra	---	------	----	111
Trifolium wormskjoldii	---	------	----	111
Vicia sativa	---	------	----	-11
Galium aparine	---	------	----	1-1
Achillea millefolium	---	------	----	1--
Holcus lanatus	---	------	----	1--
Lotus corniculatus	---	------	----	--1

SSM Successional Salt Marsh Type
MHSM Mature High Salt Marsh Type
TSM Transitional Salt Marsh Type

Tidal Creeks as Structural Controls

While elevation is a primary control on wetland hydrology, tidal circulation is also critical to salt marsh structure. Tidal creeks, in particular, are a major determinant of marsh hydrology. Over the eleven years of marsh restoration, creek depths increased and channel widths narrowed. This pattern varied, however, with the specific hydrologic connection between the creek and Salmon River. In 1979, of the eight investigated creek systems (Figure 5), Creeks 1, 2, 4, and 8 had reasonably good connection with the river. Other study site creeks (3, 5, 6, 7) had impeded hydrological connection with the river. We noted that creeks not only deepened due to erosional down-cutting but also because sedimentation elevated the marsh surface relative to the creek bed. The complexities in development of hydrological connection between river and marsh are best discussed on a creek-by-creek basis for selected creek groups.

Creek 1. The longest, deepest and most sinuous tidal creek in the restored marsh, Creek 1 was sampled at nine cross-sections. Creek 1 traverses an area of fine textured silts and was dominated by C. lyngbyei cover. Tidal connection was relatively unimpeded, except where the channel narrowed at the breached dike. Figure 17 shows four representative Creek 1 cross-sections. All nine cross-sections exhibit creek deepening and narrowing. The lower reach of the creek became deeper and narrower because of down-cutting; the middle and upper reaches became narrower by both down-cutting and lateral sedimentation. Creek deepening was greatest (55 cm) near the Salmon River

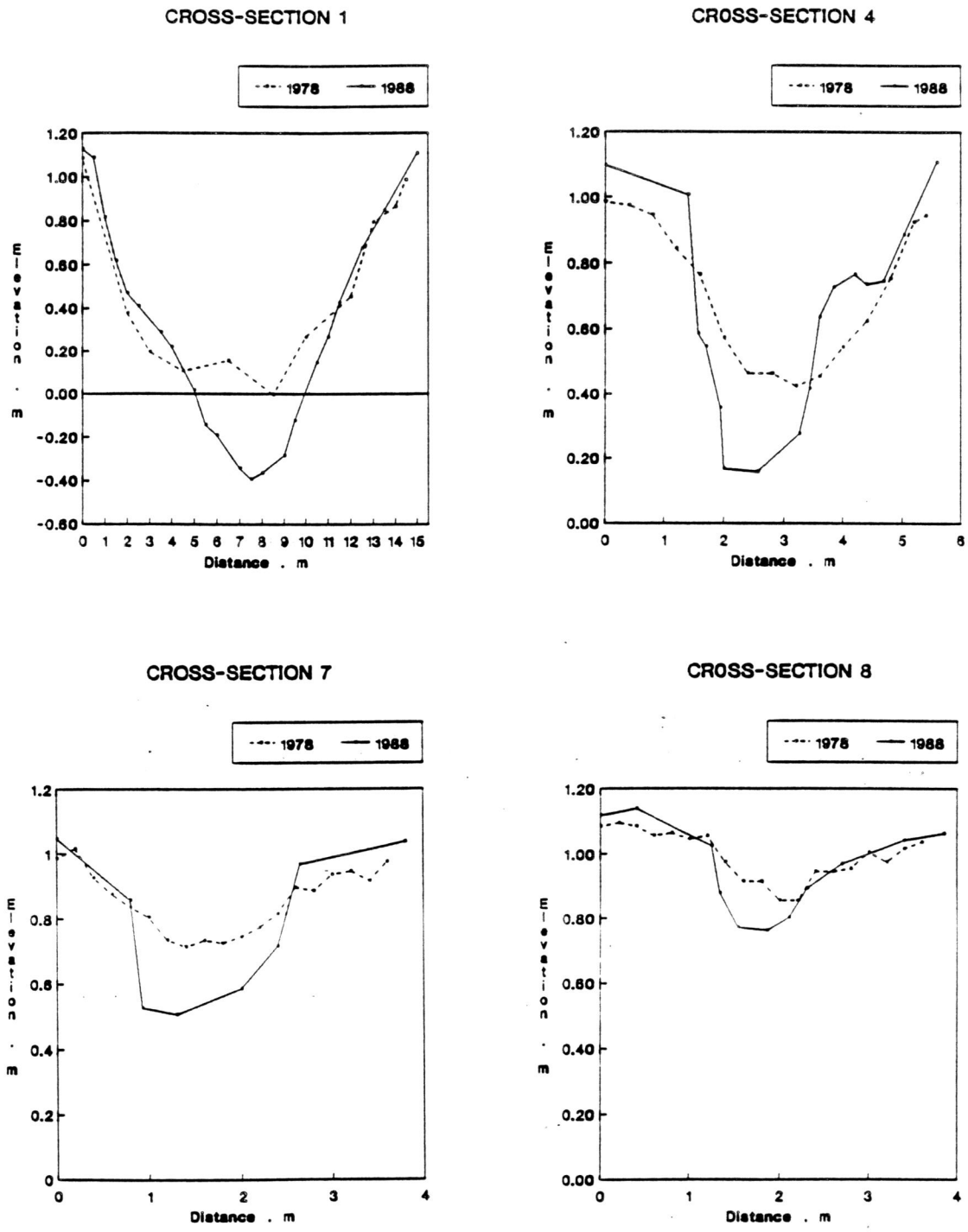


Figure 17. Representative creek cross-sections along Creek 1 based on elevation surveys in 1978 and 1988 in the north shore study site, Salmon River estuary.

(Figure 17) and least (5 cm) at the creek head. Likewise, the volume of material eroded by the creek, was also greater near the mouth than head. Figure 18 shows over-steepening of the creek profile at the mouth (cross-section 1) and suggests that in the future the creek will erode headward as it reestablishes equilibrium at grade.

Creeks 2, 4, 8. These creeks, unblocked by remnant dikes for eleven years, had reasonably good connection with the river. However, none were particularly well developed prior to diking 27 years ago. All show similar characteristics: slight deepening in the lower reach; channel narrowing in the middle reaches by both down-cutting and lateral sedimentation; and little change in cross-section or sedimentation in the upper reach. Figure 19 shows typical cross-sections.

Creek 5. This prominent creek is a broad L-shaped slough with no tributary drainages and is anomalous with respect to the other creeks in its development since dike breaching in 1978. At the time of breaching, the slough was ponded behind the dike. The lower and middle reaches show sedimentation with slight incision into the recently deposited sediments. The upper reaches show little to slight deepening. It appears that such a broad channel (15-45 m wide) is not appreciably deepened by tidal hydraulics over the eleven year restoration period; furthermore, unlike other creeks with active water circulation, the broad channel is densely covered by *C lyngbyei* in a number of places. Figure 20 show typical cross-sections.

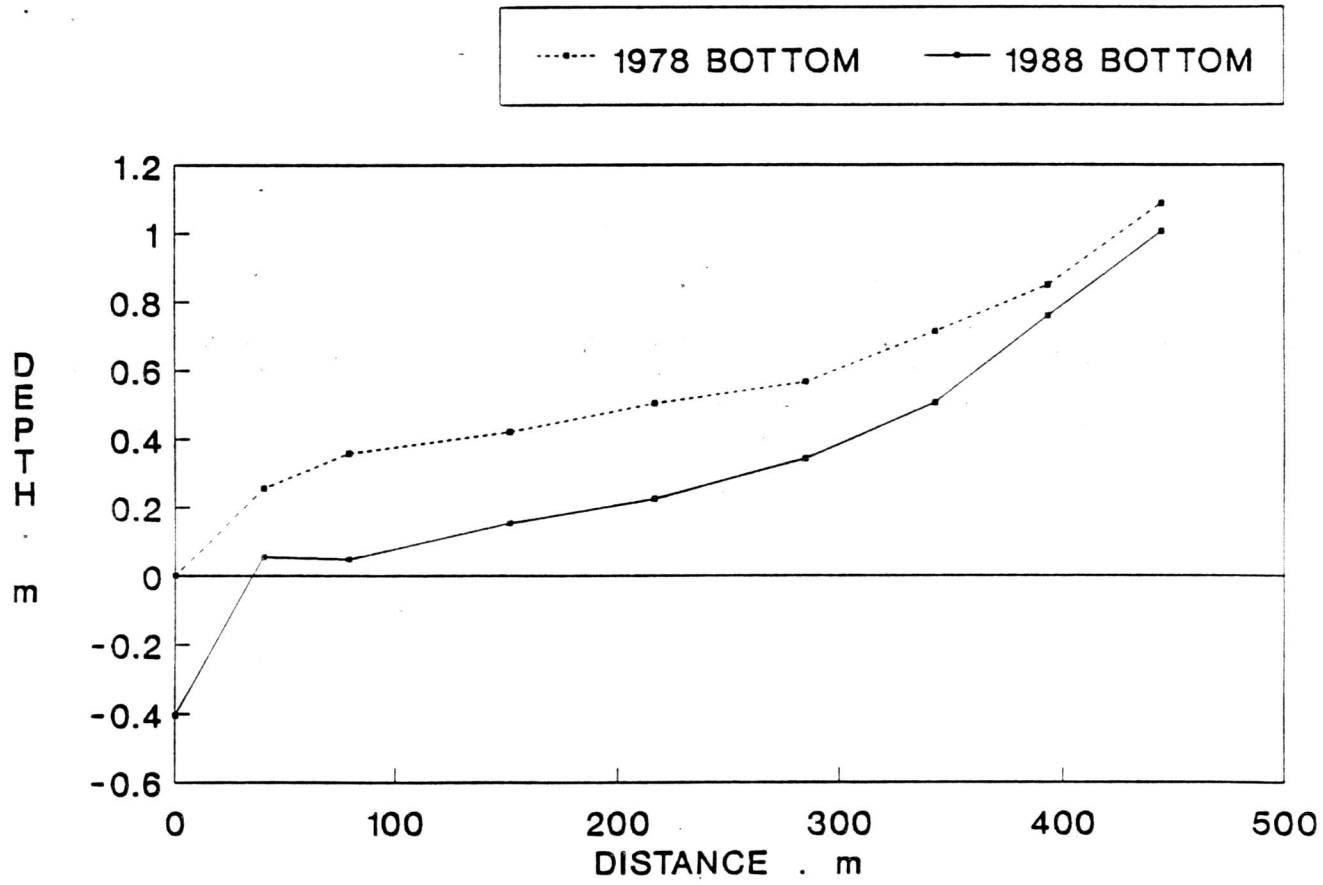


Figure 18. Elevation profile along Creek 1 in the north shore study site, Salmon River estuary based on surveys in 1978 and 1988 at nine cross-sections.

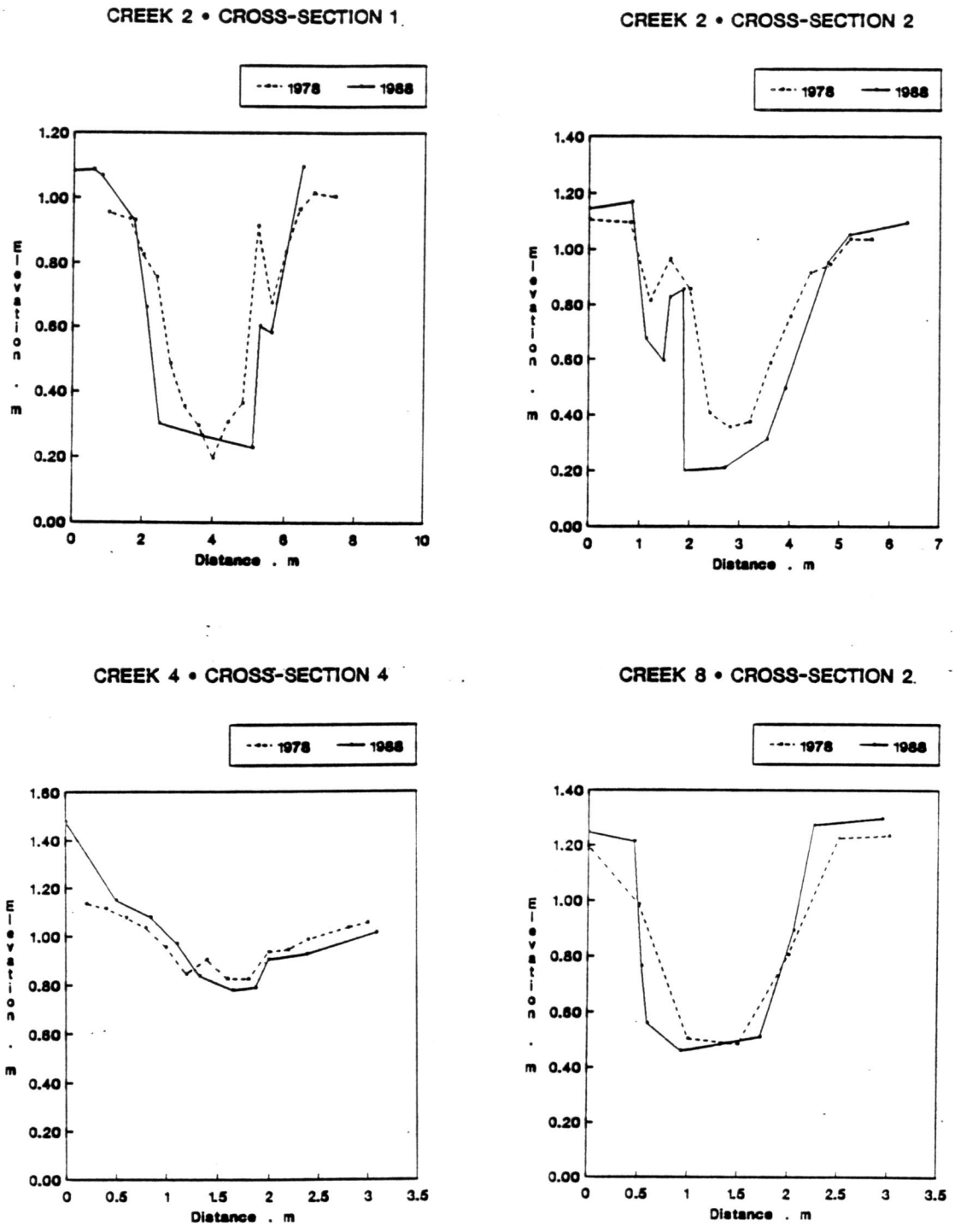


Figure 19. Typical creek cross-sections along Creeks 2, 4 and 8 based on elevation surveys in 1978 and 1988 in the north shore study site, Salmon River estuary.

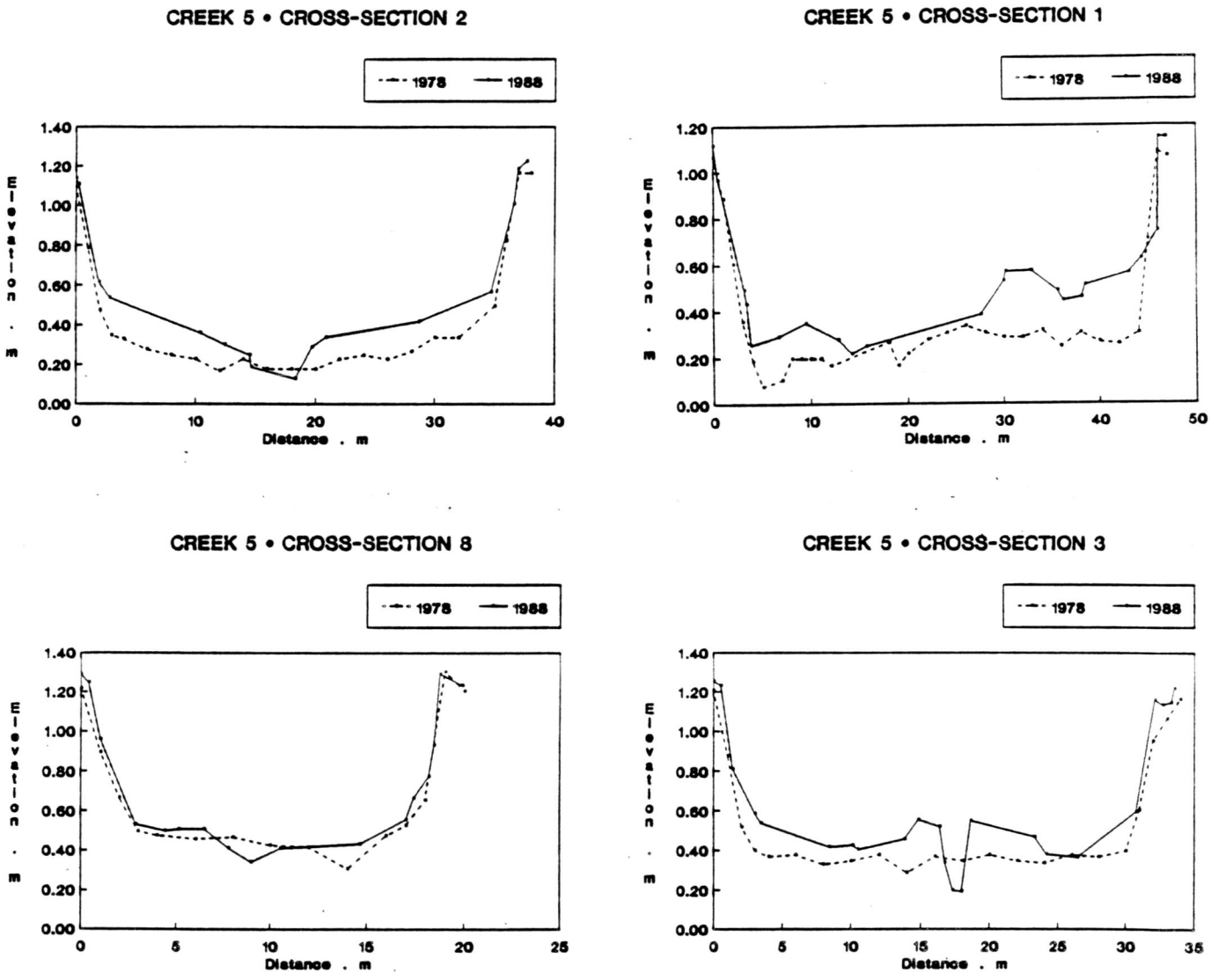


Figure 20. Typical creek cross-sections along Creek 5 based on elevation surveys in 1978 and 1988 in the north shore study site, Salmon River estuary.

Creeks 6 and 7. Post restoration cross-sections of these two creeks are particularly instructive since in 1978, connection of these creeks to the Salmon River was primarily an excavated narrow slot in the still-intact dike. A 200 m long section of dike remained until September 1987 when it was leveled by an excavator (Figure 5). The presence of the dike segment, despite the narrow creek connection, inhibited tidal exchange. In the case of Creek 7, there was very slight down-cutting in the lower reach and 15 cm of sedimentation in the upper reach (Figure 21). Creek 6, a very short drainage, showed sedimentation of 30 cm and no down-cutting. Much of the cross-sections of these creeks, especially in their upper reaches was completely covered by marsh vegetation. Experience with these two creeks suggests the importance of full dike removal for a successful restoration.

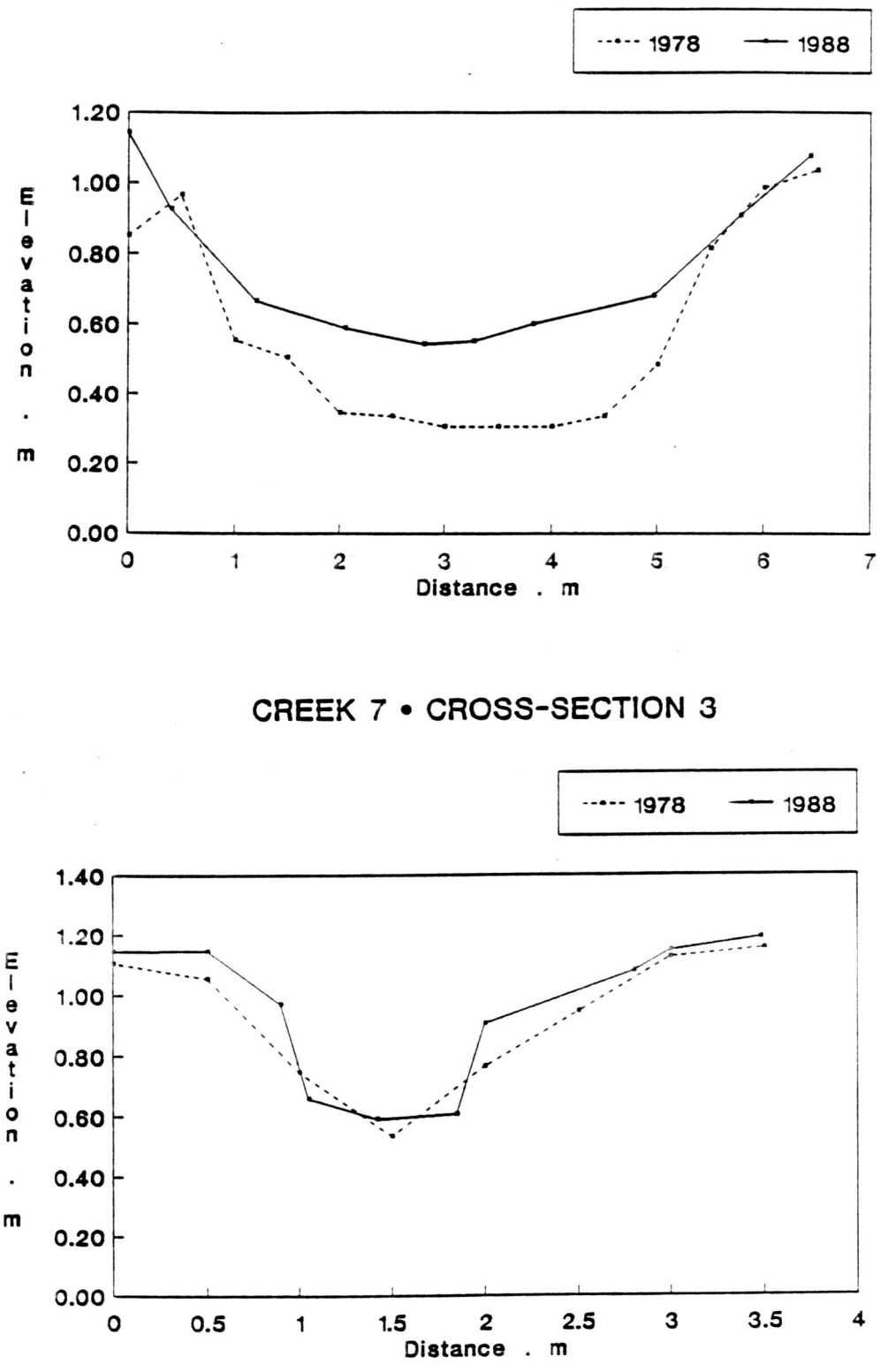


Figure 21. Typical creek cross-sections along Creeks 6 and 8 based on elevation surveys in 1978 and 1988 in the north shore study site, Salmon River estuary.

Soils, Substrates and Wetland Structure

Soils and substrates result from both marsh development and help direct development. Where substrates are sandy, drainage is enhanced. Where substrates accrete by vegetation trapping sediments, marshes build up and support high marsh species rather than low. Soils rich in organic matter may develop from local litter decomposition. In 1988, we assessed a number of soil and substrate properties at each permanent plot. Mitchell (1981) also investigated soil properties in the north shore study site from 1978-1980 based on four composite soil samples, two in the restoration area and two in the controls. Measurements were made on soil chemistry, including CEC, organic matter, NH_4-N, NO_3-N, P, K, Ca, Mg, Na, and B. She also described marsh soil profiles in the standard way. Additionally, she annually extracted a core from each permanent plot which was used to measure interstitial salinity and pH. We did not conduct as detailed a study of soil chemistry as did Mitchell.

In this section we address four soils characteristics: sediment accretion, soil texture, organic content, and salinity. In part, we relate these characteristics to elevation and floristics and discuss the changes since 1978.

Sediment accretion. By accretion we mean substrate addition including additions of inorganic and organic sediments. The latter may be transported by tidal waters or deposited locally by plant litter accumulation and decomposition. In 1978, Mitchell

placed a 0.5 cm-thick layer of sand on the pasture surface at each permanent plot. This sand layer tagged the wetland surface at the beginning of the restoration period. In 1979, she observed that a thin layer of sediment had already covered the sand tag. In 1988, we cored through the sand tag and measured the thickness of sediment above the tag.

Mean sediment accretion in the restoration area was 3.6 cm (s.d 1.44, s.e. 0.22) as shown in Table 14. Maximum accretion was 7.7 cm, minimum 1.4 cm. It is clear that there was much variability in accreted sediment thickness. Unfortunately, we were unable to relocate sand tags in control Area 50 and could not compare accretion in the control with the restoration area. All plots showed accretion over the restoration period. Figure 22 maps the sediment accretion in the restoration area. Maximum accretion occurred in low areas dominated by Carex lyngbyei (values in excess of 5 cm in eleven years). There was a tendency for accretion to diminish higher in the marsh. We interpret this general pattern to be caused by increased tidal exchange in the Carex marsh where there were greater amounts of sediment input and increased productivity contributing organic matter build up.

Accretion regressed on elevation for all permanent plots in the restoration site exhibits a very weak negative correlation ($r=0.20$, $p=0.20$) with much scatter. One might anticipate a stronger relation between accretion and elevation change from 1978 to 1988, since increase in elevation expressed by elevation

change would be substantially accounted for by inorganic and organic sediments. This stronger relation is shown by Figure 23 with a positive correlation of r=0.48 (p=0.001). Elevation change and accretion data approximated normal distributions. We also applied a Spearman rank correlation which does not assume normality in which the coefficient of correlation was r_S=0.51, p=0.001).

Table 14. Selected soils and environmental characteristics in 1988 in restoration area of the north shore study site, Salmon River estuary.

	Characteristic					
	Accret. (cm)	Sand Content (%)	Organ. Content (%)	Salin. (ppt)	Elevat. (m NGVD)	Elevat. Change (cm)
n	44	46	48	47	49	49
Mean	3.57	2.64	36.9	31.5	1.23	5.12
Std. Dev.	1.44	2.73	10.6	5.6	0.10	2.42
Std. Error.	0.22	0.40	1.5	0.8	0.01	0.35
Minimum	1.40	0.00	15.4	22.0	1.06	-2.30
Maximum	7.70	13.50	62.9	44.0	1.55	9.90

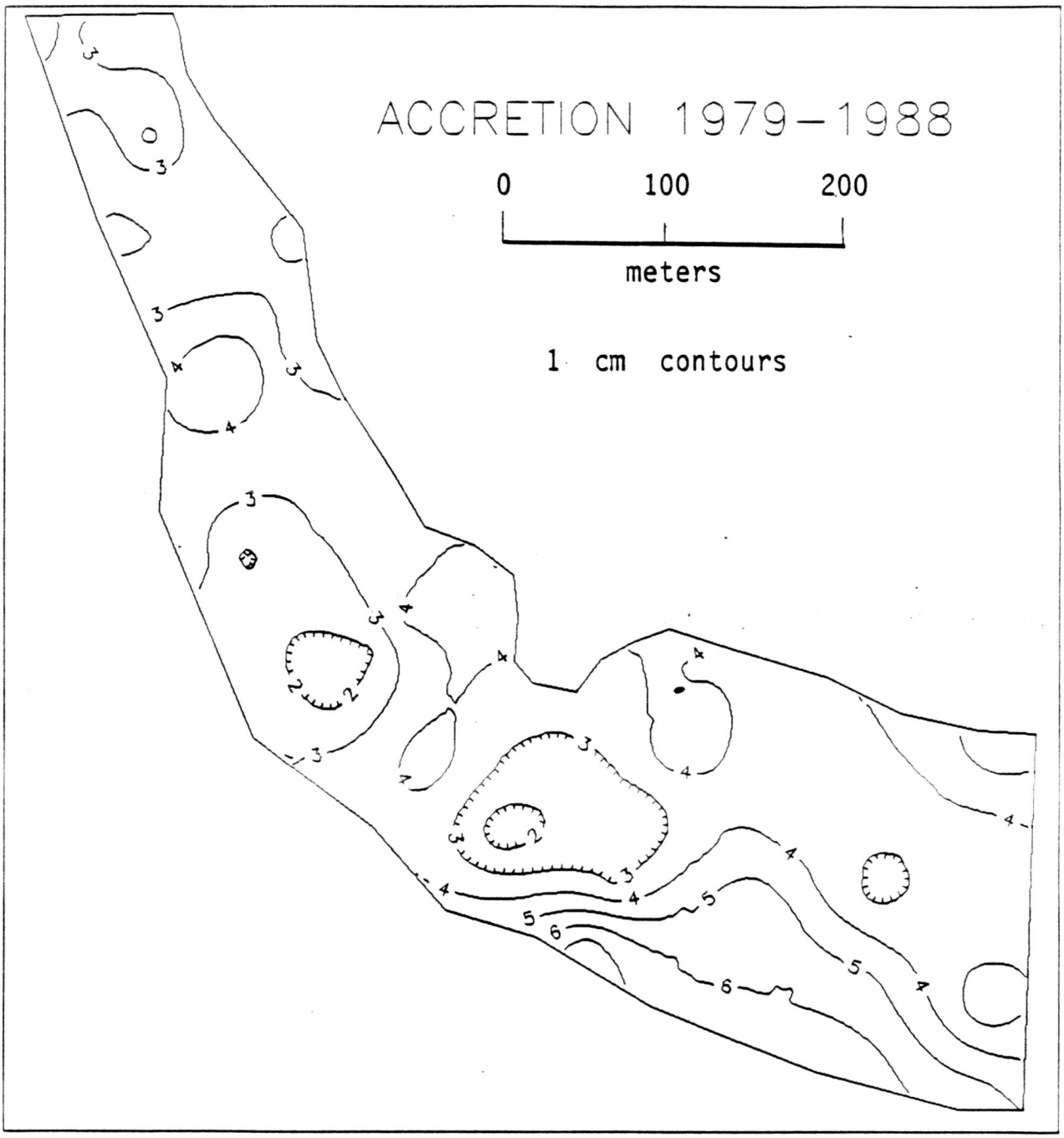


Figure 22. Sediment accretion above a sand tag in the Salmon River estuary north shore restoration site from 1979 to 1988.

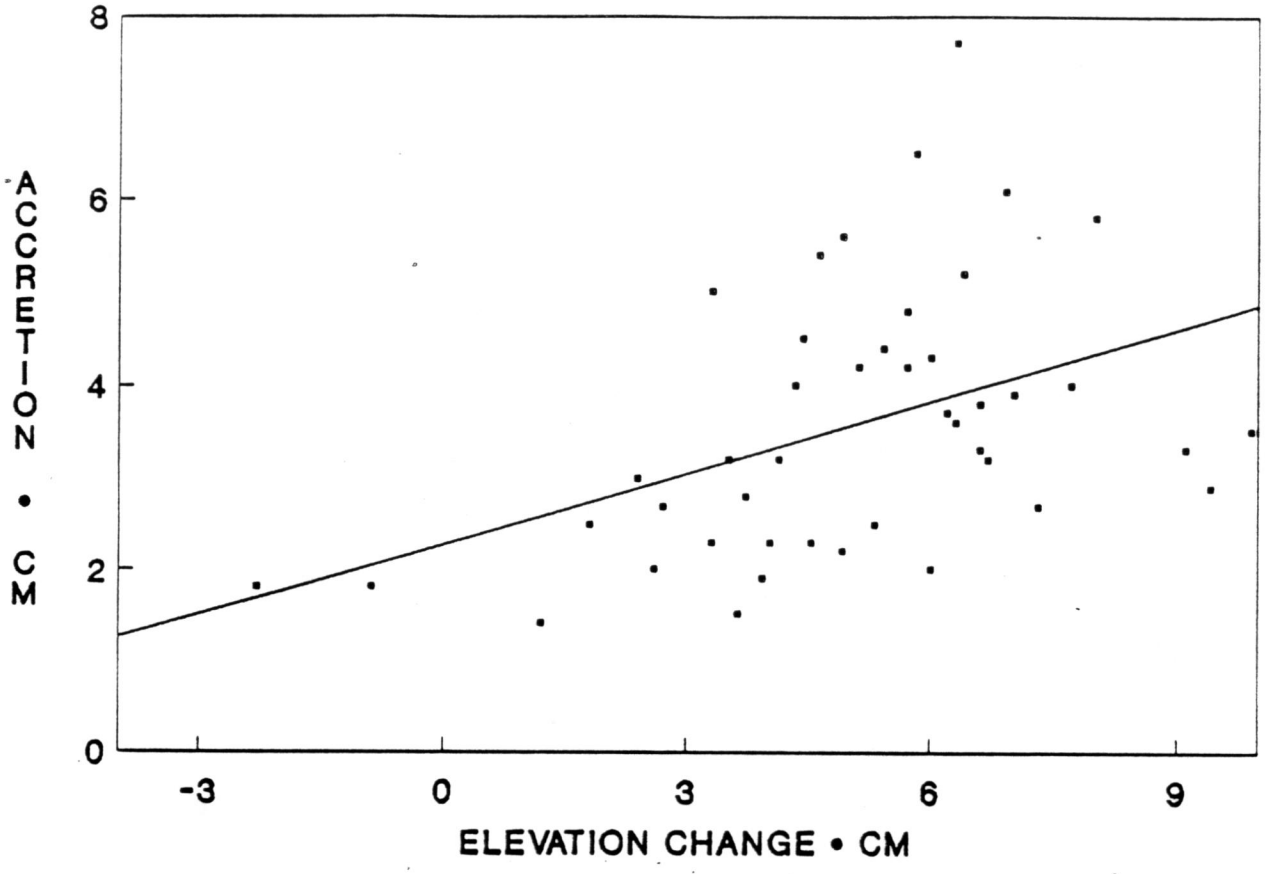


Figure 23. Relation between sediment accretion and elevation change from 1979 to 1988 in the Salmon River north shore restoration site based on permanent plot data, r=0.48, p=0.001.

Substrate texture. Mitchell (1981) did not record soil texture. We observed that soils downriver, in the north part of the restoration area and control Area 53, were sandier than upriver, in the south. This observation was consistent with the dominance of Salicornia and Distichlis downriver and Carex upriver. It was also consistent with the estuarine position of the more northern part of the site which was exposed to the mouth of the estuary and sandbar. On the other hand, the more southern parts of the study site were exposed to siltier riverine sediments rather than marine sands.

Soil texture was separated into sand and silt/clay-size fractions. Figure 24 maps the distribution of percent sand in the north shore restoration site confirming our superficial observation. Sandy soils appear to be associated with Salicornia and Distichlis cover. Since neither the percent sand nor percent species cover met tests of normality, we evaluated the strength of the relation between the Salicornia cover and percent sand by Spearman's rank correlation which gave a coefficient of association of r_S=0.48, p=001; for the relation between Distichlis and sand r_S=0.56, p=000; and, percent Carex cover was **negatively** related to percent sand, r_S=-0.42, p=005.

Another property of the restored area soils was the percent organic matter vs. inorganic matter. Figure 25 displays a map of percent soil organic matter (loss on ignition) in the north shore restoration area based on soil cores extracted from 49 permanent plots. The map shows organic matter increasing toward the upland. Figure 26 shows a moderately strong (r=55, p=.000) correlation between percent soil organic matter and elevation. Higher contents of inorganic sediments appear to be associated with greater tidal exchange as shown by the correspondence of low organic content with the position of major creeks, Creek 1 and Creek 5 (compare Figure 5 with Figure 25).

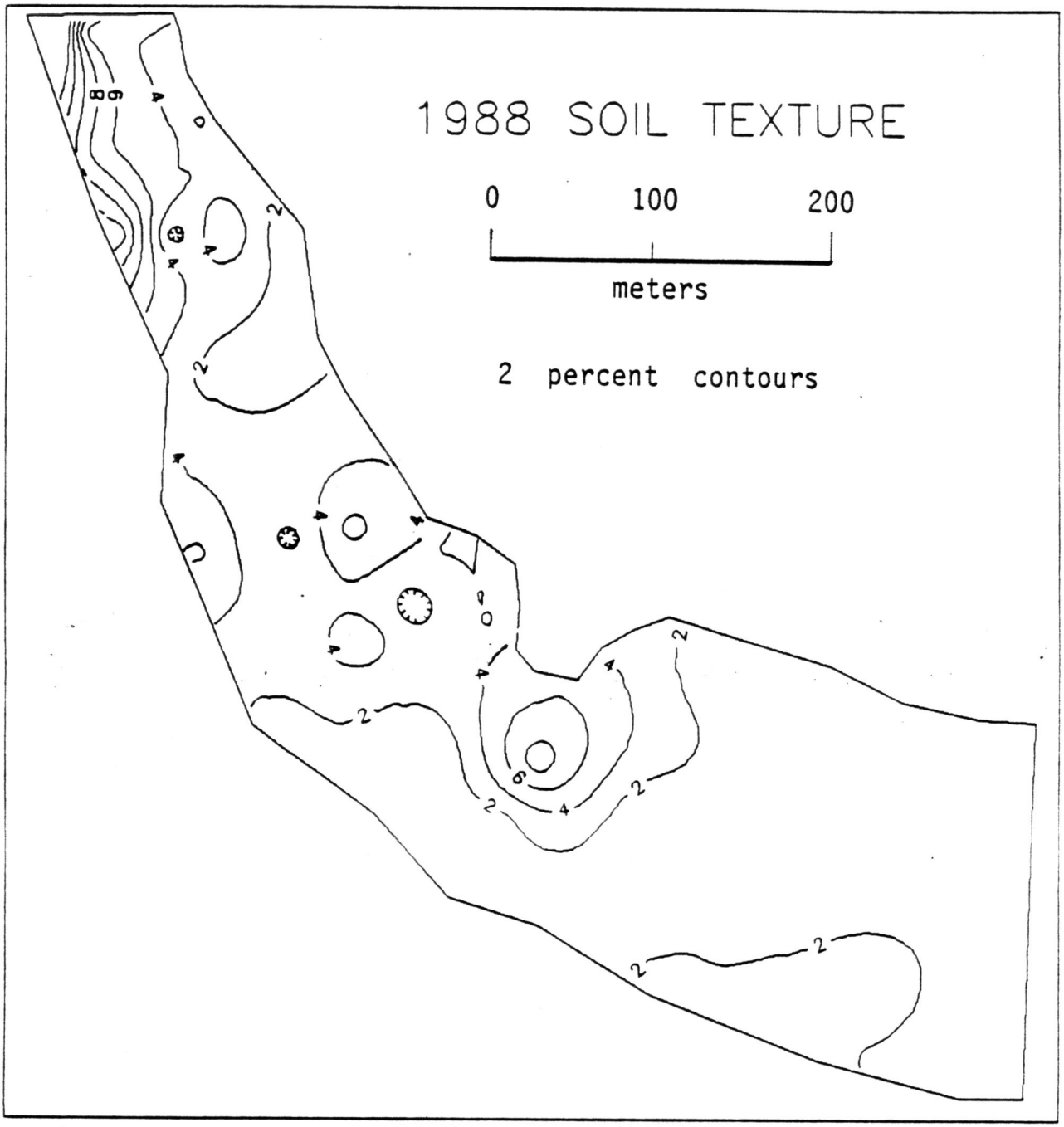


Figure 24. Percent sand in soil samples in the Salmon River estuary north shore restoration area in 1988, an expression of soil texture.

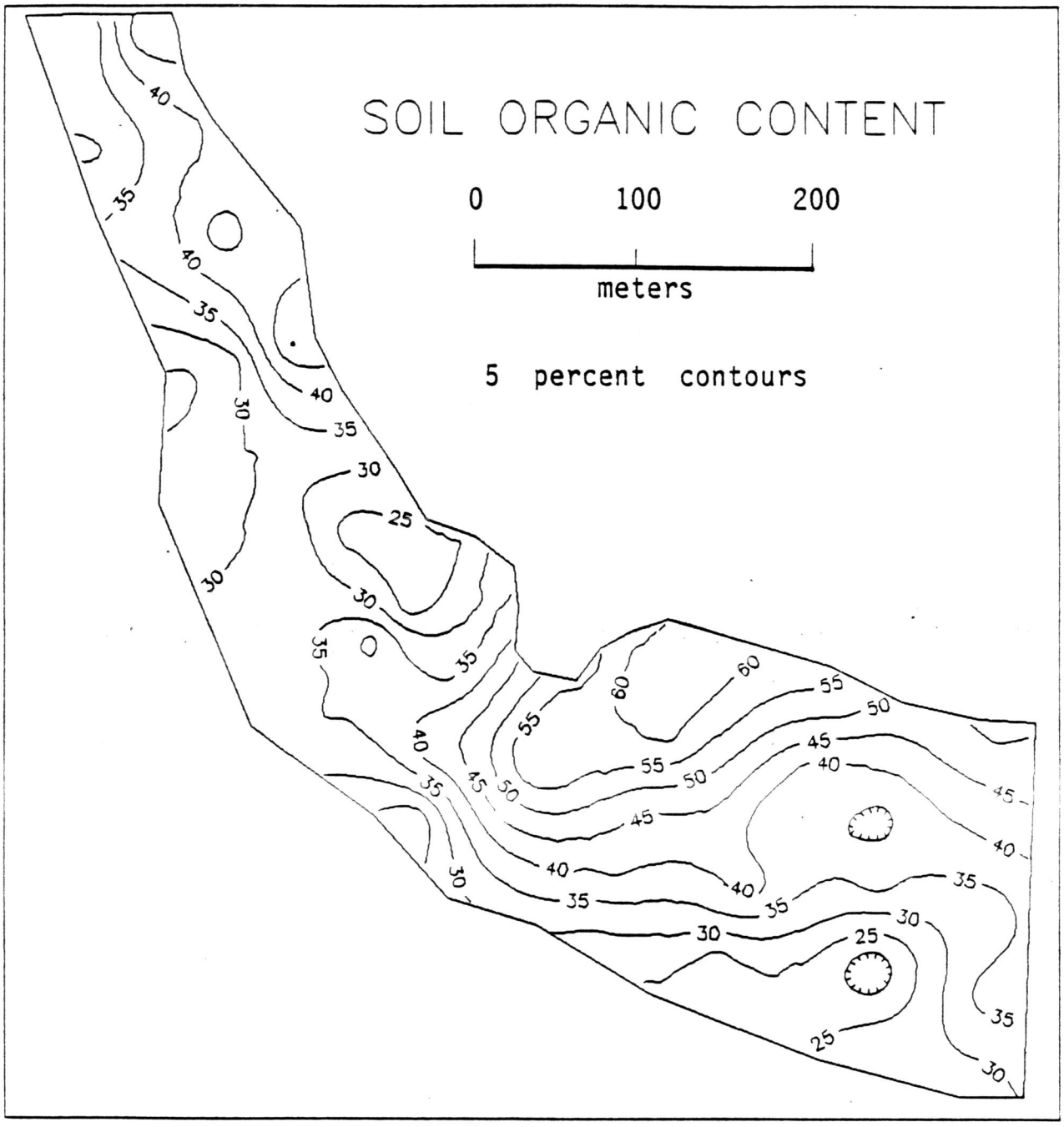


Figure 25. Percent organic content in soil samples in the Salmon River estuary north shore restoration area in 1988, based on loss on ignition from 49 permanent plots samples.

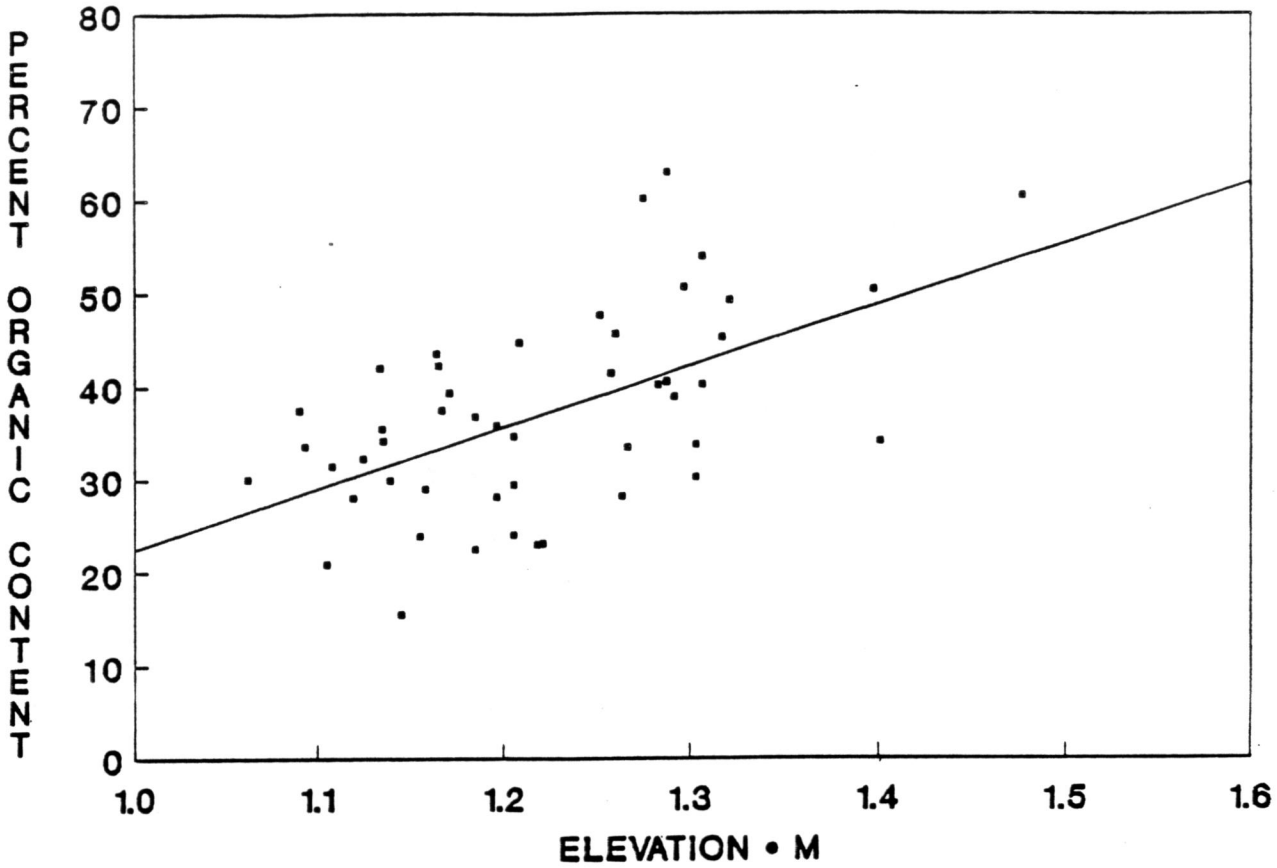


Figure 26. Relation between percent loss on ignition organic content of soil and elevation (m NGVD) in 1988 in the Salmon River north shore restoration site based on permanent plot data, r=0.55, p=0.000.

Salinity. Interstitial soil water salinity was close to 0 ppt in 1978 over most of the restoration site. Mitchell (1981) measured salinities at all permanent plots in the restoration site and at selected plots in the control areas on 12 September 1979 and 30 September 1980. In the two years after breaching, mean salinities ranged from 18 to 33 ppt with highest salinities in low elevation areas with much bare soil. Salinities in the control areas were lower (18 to 25 ppt in Area 50, upstream, and 3 to 37 ppt in Area 53, downstream). Control Area 53, with many enclosed ponded depressions, had some very high salinities.

On 7-8 September 1988 mean salinity in the restoration area was 31.5 ppt with a 22 to 44 ppt range (Table 15). Figure 27 (next page) maps the distribution of salinities and shows the expected higher salinities (> 33 ppt) downriver in the northern part of the restoration area where Salicornia and Distichlis communities prevail on more sandy substrate. Lower salinities (<32 ppt) characterize the upstream area with Carex. There was a moderate positive relation between salinity and percent sand ($r=0.51$, $p=0.000$; $r_s=0.51$, $p=0.000$) as shown in Figure 28.

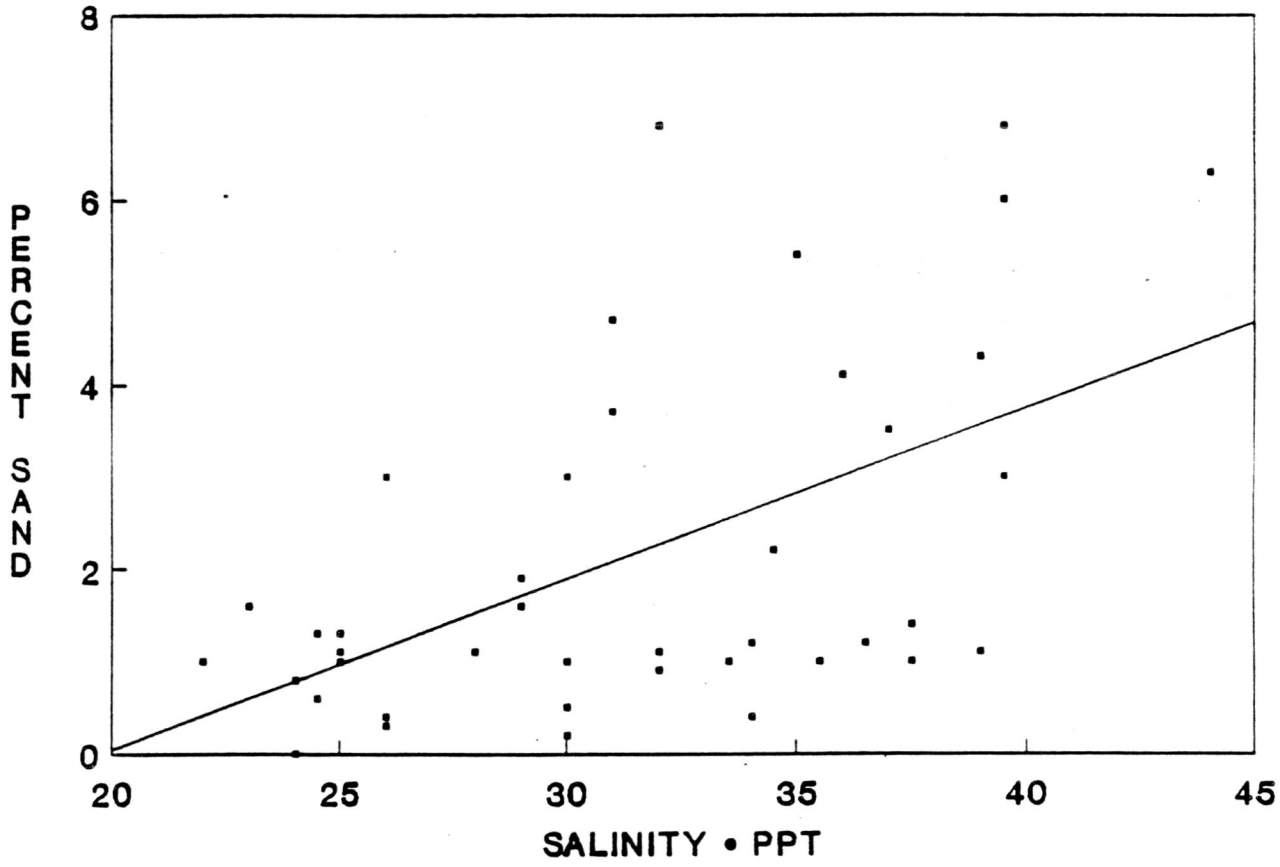


Figure 28. Relation between percent sand content of soil and interstitial soil water salinity (ppt) in 1988 in the Salmon River north shore restoration site based on permanent plot data, r=0.51, p=0.000.

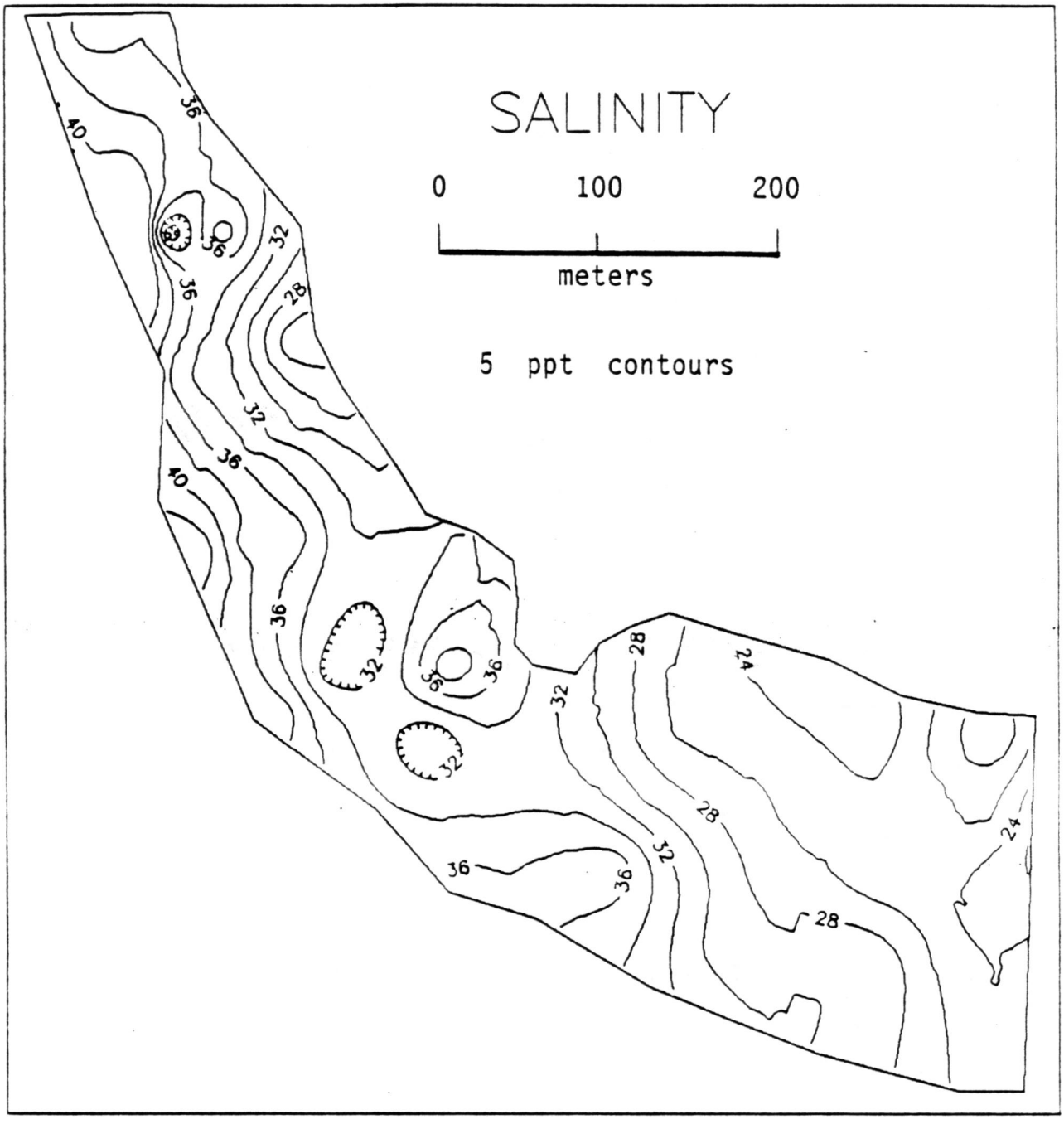


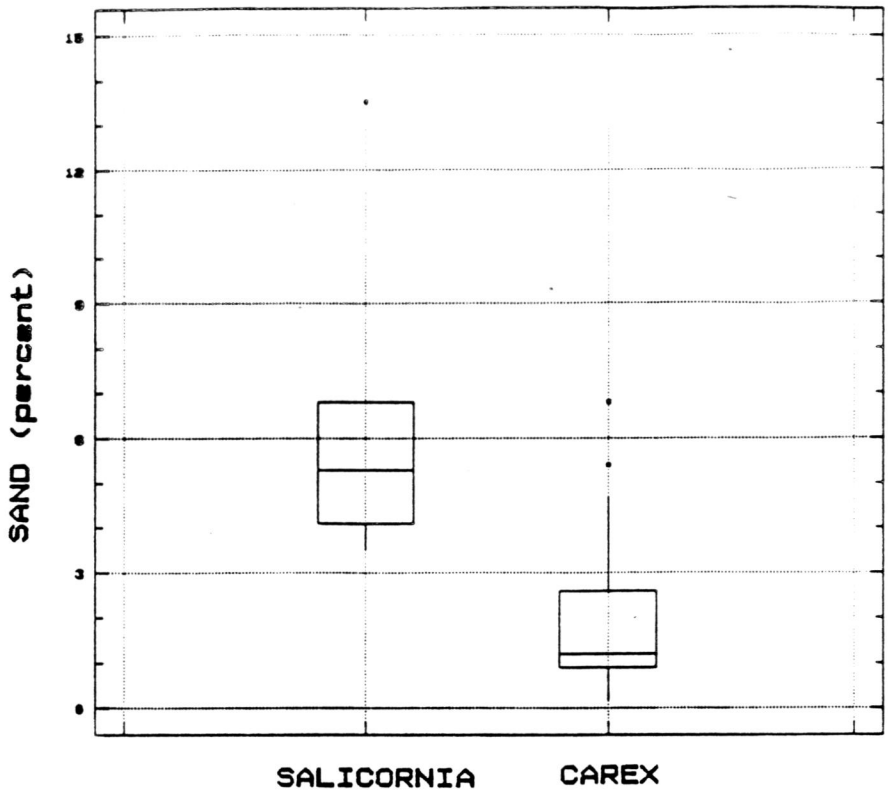
Figure 27 . Interstitial soil water salinity (ppt) in soil samples in the Salmon River estuary north shore restoration area in 1988, based on 49 permanent plots samples.

Species distribution was also related to salinity. Because normality assumptions could not be met, we applied Spearman's rank correlation to test the association of dominant species with salinity. Two species exhibited moderately strong positive associations with salinity: Distichlis with r_S=0.64, p=.000 and Salicornia with r_S=0.58, p=.000. Carex, however, was significantly negatively associated with salinity (r_S=-0.39, p=0.008). These associations were in accord with the downriver position of the area dominated by Salicornia and Distichlis and with the distribution of sandy soils.

Plant community-soil associations. Mitchell (1981) described variation in pasture communities and attributed differences, in part, to different land use histories. Since restoration in 1978, distinctive low salt marsh plant communities have begun to develop (Figure 8, p. 49). The environmental structure of these communities varies considerably. Although there there were differences in soil propeties between various communities, we did not consider our sample sufficiently large to characterize many of the communities; therefore, we chose to aggregate permanent plots into two community groups: a Salicornia-Distichlis-Triglochin-Juncus Group and a Carex Group. This community dichotomy is distinguished by the soil characteristics shown in Table 15. The groups are not significantly different with respect to elevation, elevation change, percent organic content, and accretion (Student's t test and Mann-Whitney U-test). The communities are significantly different with respect to percent sand and salinity (Table 15 and Figure 29).

Table 15. Selected soils and environmental characteristics and their differences for two community groups representing 76 % of the permanent plots in the Salmon River estuary north shore restoration area in 1988. Student's t-test and Mann-Whitney Z tests are used to compare communities with respect to a given property (** = significant at p=0.01).

	Characteristic					
	Salin. (ppt)	Sand (%)	Organ. (%)	Accret. (cm)	Elevat. (m NGVD)	Elevat. change (cm)
SALICORNIA-DISTICHLIS-TRIGLOCHIN-JUNCUS GROUP						
n	7	6	8	7	8	8
mean	38.6	6.4	35.4	2.9	1.22	3.8
std. dev.	2.8	3.7	6.8	2.7	0.09	3.7
std. err.	1.1	1.5	2.4	1.5	0.03	1.3
min.	36.0	3.5	28.1	1.8	1.06	-2.3
max.	44.0	13.5	49.0	4.8	1.32	9.1
CAREX GROUP						
n	29	28	29	27	29	29
mean	30.0	1.9	34.8	3.7	1.20	5.5
std. dev.	4.4	1.7	9.7	1.6	0.07	2.0
std. err.	0.8	0.3	1.8	0.3	0.01	0.4
min.	23.0	0.2	15.4	1.4	1.09	1.2
max.	37.5	6.8	60.0	7.7	1.40	9.9
COMPARISON						
t-test	4.96**	4.79**	0.15	1.19	0.77	1.67
Mann-Whitney Z	-3.72**	-3.21**	0.00	-1.21	-1.09	1.05



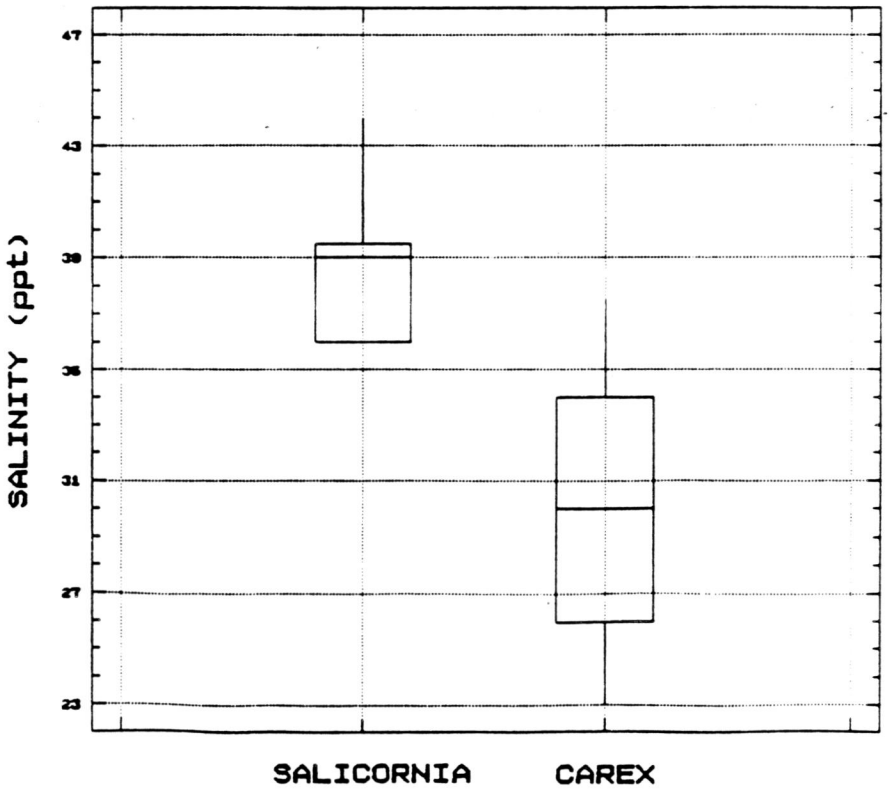


Figure 29. Comparative distribution of sand content (%) and salinity (ppt) for the Salicornia Community Group (n=8) and Carex Community Group (n=29) in the restoration area of the Salmon River estuary north shore study site in 1988.

The *Salicornia* Group had a mean salinity of almost 39 ppt (mid-September measurement) and a mean sand content of 6.4 % compared with the *Carex* Group with 30 ppt salinity and 1.9 % sand. There appeared to be greater accretion and greater elevation change for the *Carex* Group than for the *Salicornia* Group but because of the small n size and scatter in the data, these differences were not statistically significant.

Function of the North Shore Study Site

By function we mean the services provided by an ecosystem. Among frequently cited wetland ecosystem services are: flood protection, primary production providing food for fish and wildlife, wildlife and fisheries habitat, recreation, and education and research. Additionally, ecologists frequently regard nutrient cycling and diversity as functional components of ecosystems. We have only compiled data on primary production based on peak biomass and vascular plant species diversity. In the future it would be possible to evaluate a number of other services provided through the restoration of the Salmon River salt marshes.

Biomass

Peak standing biomass can be an important indicator of salt marsh species and community above ground net primary production (Kibby et al. 1980). Mitchell harvested peak biomass from 0.1 m^2 plots at permanent plot locations in the restored area in 1978, 1979, 1980, 1981, 1982, and 1984 and reported summary data for 1978-1980 (Mitchell 1981). In 1988 we harvested biomass in 49 restored area plots. In the following discussion, we trace changes in biomass since dike breaching in the restored and control areas. Biomass, expressed below as g/m^2 dry matter, was harvested, sorted by species, dried, and weighed. Biomass was harvested at all permanent plots in the restored area and in control plots along two transects in years up to 1984.

Figure 30 and Table 16 show trends in total biomass in the north shore study areas from 1978 to 1988. Figure 31 displays a statistical distribution of biomass production for the restoration area for 1980, 1984, and 1988. In the restoration area, peak biomass diminished from about 750 g/m^2 in 1978 when the area supported pasture to about 200 g/m^2 in 1980 when substantial bare ground was exposed by mortality of pasture species prior to expansion of colonizers. Biomass rapidly increased from 1980 to 1988 when it reached about 1650 g/m^2, more than twice the biomass of the control and original pasture.

In the the control area, total biomass fluctuated with an average of about 500 g/m^2. The higher biomass value in 1978 of 826 g/m^2 in the control area (Table 16) represents five plots in Area 51 while in other years biomass was collected from 11 plots in both Area 50 and 53. We consider the data reported for 1979-1984 to be more representative of the control.

Changes in individual species biomass follow those already described for changes in species cover. Table 17 show changes in major species in the restoration area from 1980 to 1988. Mean biomass of Agrostis alba and Potentilla pacifica, both residual from the pasture, varied little from year to year, Agrostis having about four times the biomass of Potentilla. Major colonizing species: Carex lyngbyei, Distichlis spicata, Juncus balticus, Salicornia virginica, and Triglochin maritimum, all increased. Biomass of Carex, Distichlis and Salicornia increased strikingly.

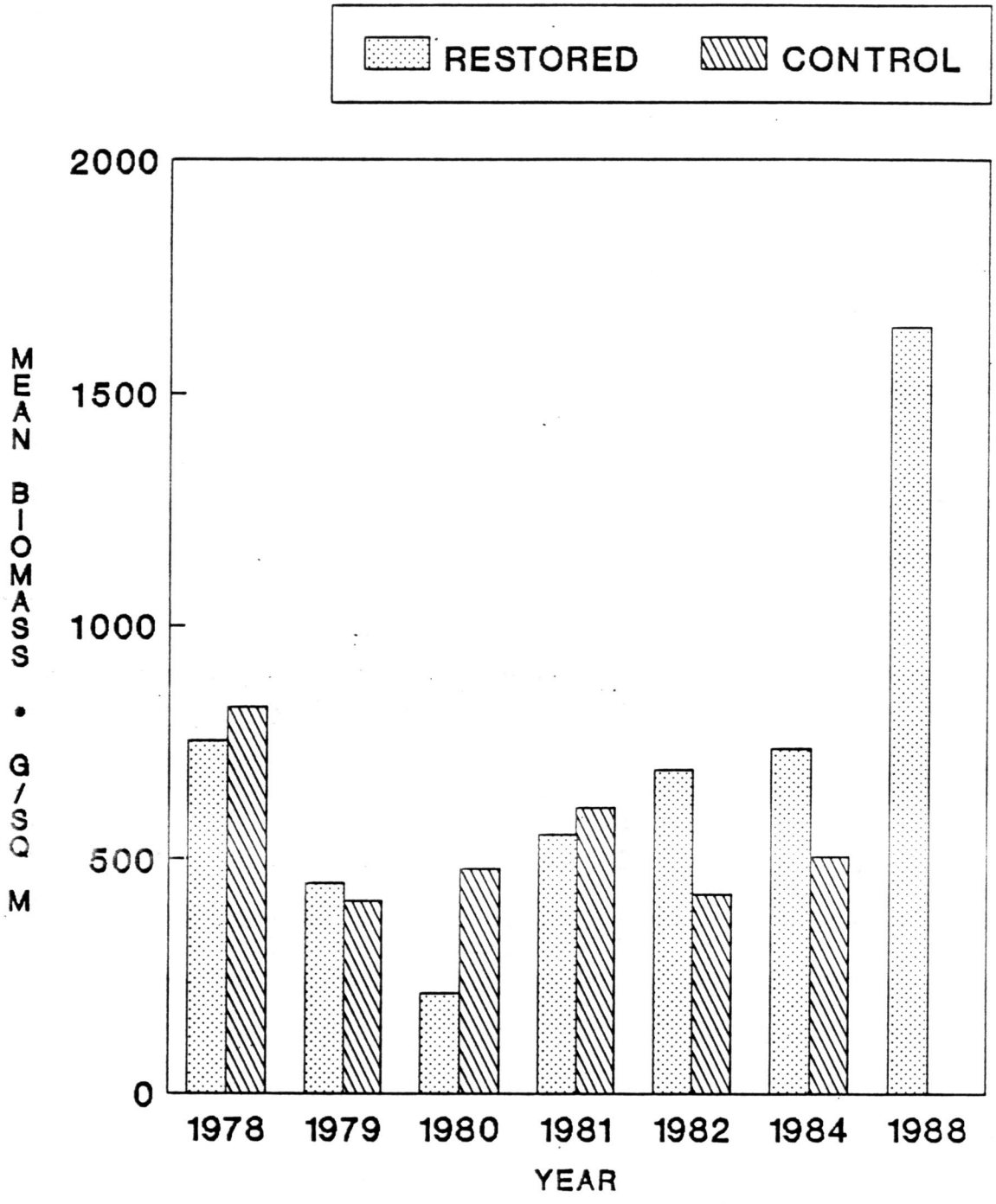


Figure 30. Trends in live biomass production from 1978 to 1988 in the restored and control areas of the north shore study site, Salmon River estuary.

Table 16. Average total above ground dry biomass in g/m^2 by year in the Salmon River north shore study site harvested at peak biomass development.

Year	1978	1979	1980	1981	1982	1984	1988
			RESTORED AREA				
Average	756	448	215	553	693	740	1645
No. Plots	49	49	49	49	49	47	49
Std. Dev.	273	438	279	na	472	365	331
Std. Error	39	63	40	76	67	53	47
			CONTROL AREA				
Average	826	411	479	612	424	506	---
No. Plots	5	11	11	11	11	11	---
Std. Dev.	138	103	327	296	182	253	---
Std. Error	62	33	99	89	55	76	---

Biomass distribution is mapped for the restoration site in 1988 in Figure 32. The highest biomass values appeared in the upper elevations of the downriver (northern) part of the restoration area and in the upper elevations of central portion of the area. There was no significant relation of biomass to any of the structural properties of the restored marsh.

We simplified the eight plant communities in the north shore study site into three generalized communities common in the restoration area: Carex community, Salicornia-Distichlis community, and Agrostis-Potentilla community. Trends in biomass

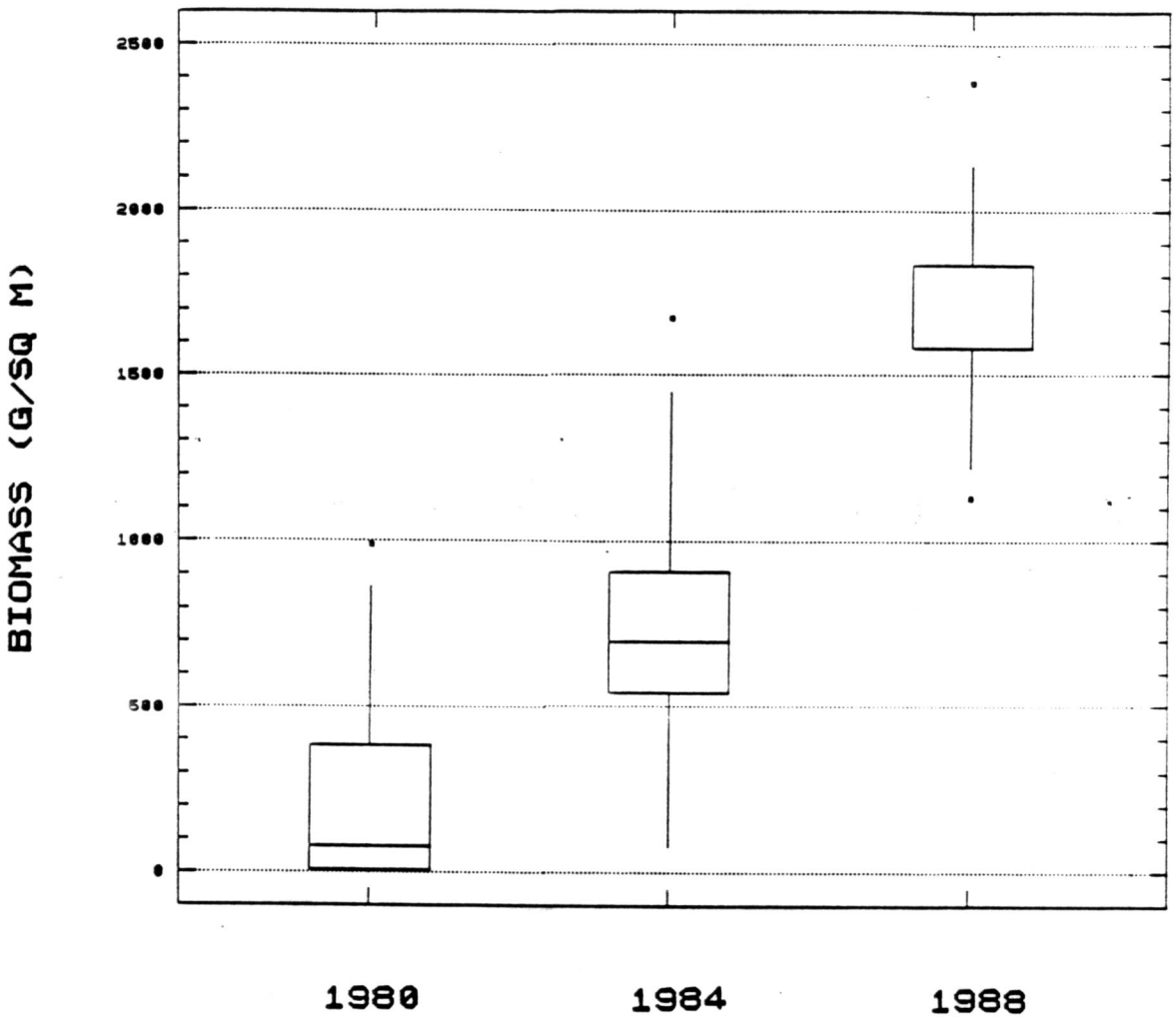


Figure 31. Statistical distribution of biomass in the restored area of the north shore study site, Salmon River estuary for 1980, 1984, and 1988.

Table 17. Mean dry weight of above ground biomass in g/m^2 of important species for 1980, 1984, and 1988 in the restoration area of the Salmon River estuary north shore study site; * = < 1 g/m^2.

SPECIES	1980 (n = 49)	1984 (n = 47)	1988 (n = 49)
Agrostis alba	126	137	146
std. dev.	(234)	(273)	(389)
std. error	(34)	(40)	(55)
Atriplex patula	7	*	7
std. dev.	(18)		(35)
std. error	(3)		(5)
Carex lyngbyei	*	431	996
std. dev.		(460)	(729)
std. error		(67)	(104)
Distichlis spicata	0	34	96
std. dev.		(113)	(219)
std. error		(17)	(32)
Hordeum brachyantherum	14	2	4
std. dev.	(60)	(13)	(26)
std. error	(9)	(2)	(4)
Juncus balticus	10	28	79
std. dev.	(51)	(92)	(189)
std. error	(7)	(14)	(27)
Potentilla pacifica	46	32	30
std. dev.	(87)	(82)	(90)
std. error	(13)	(12)	(13)
Salicornia virginica	5	49	116
std. dev.	(24)	(113)	(295)
std. error	(3)	(17)	(42)
Triglochin maritimum	*	18	35
std. error		(9)	(24)

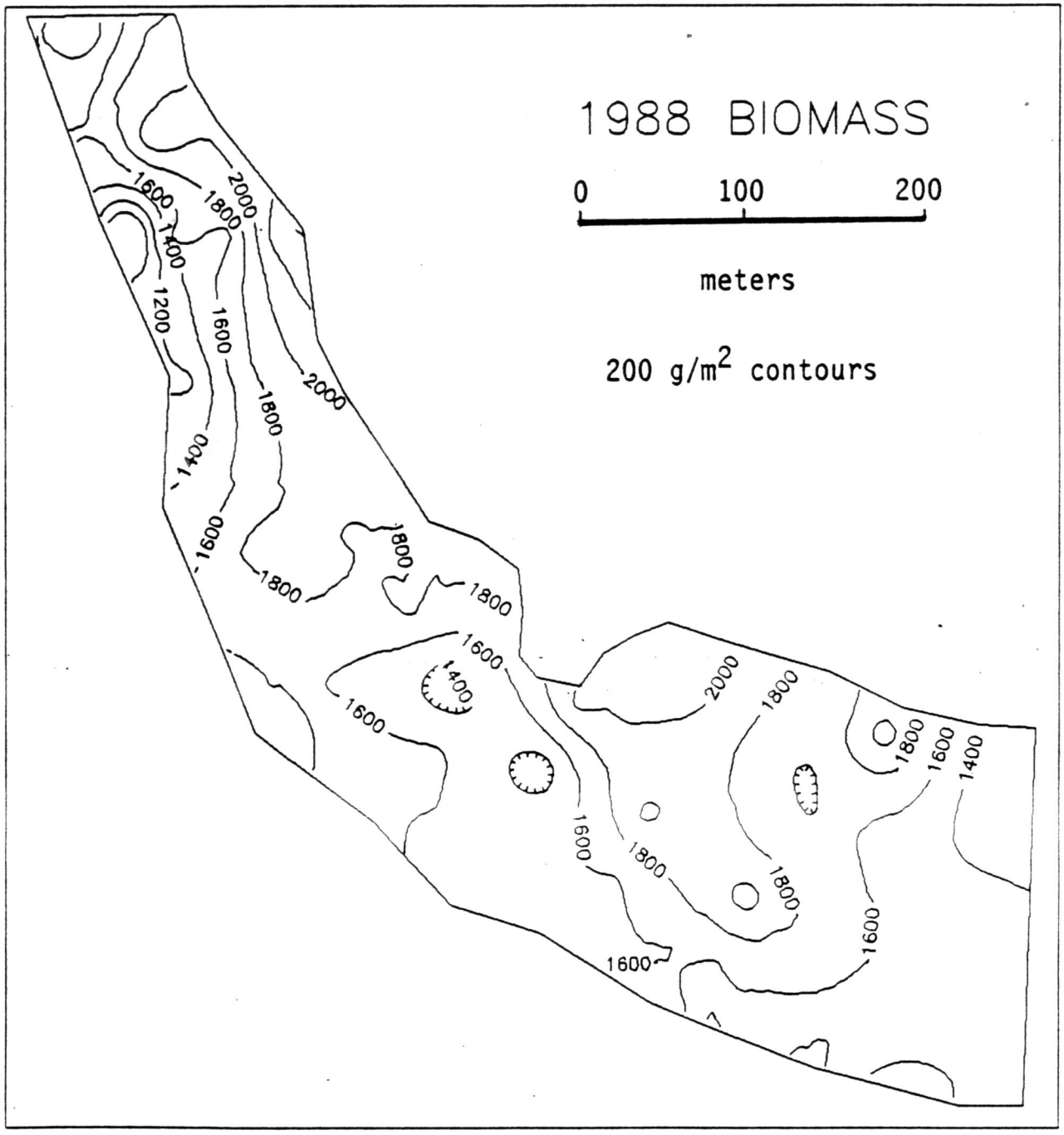


Figure 32. Biomass distribution in the restoration area of the Salmon River estuary north shore study site in 1988.

of important species in these three communities are shown in Table 18. The communities dominated by colonizing species (Carex and Salicornia-Distichlis communities) show the stronger increases in biomass than did communities dominated by residual species.

Table 18. Mean above ground dry weight biomass in g/m^2 by community for 1980, 1984 and 1988 in the restoration area of the Salmon River estuary north shore study site.

Community	No. Plots	1980	1984	1988
Carex	29	163	752	1631
std. error		44	65	40
Salicornia-Distichlis	7	114	546	1740
std. error		35	85	112
Potentilla-Agrostis	6	692	978	1786
std. error		97	158	190

Primary Productivity

We applied the same methodology as Mitchell (1981) in estimating above ground net primary productivity using the method developed by Kibby et al (1980). Their method utilizes species-specific seasonal biomass production curves from which an estimate of productivity can be made by a single biomass measurement if biomass is sorted by species. Kibby et al. (1980) developed their biomass curves from data collected at Siletz Bay and Netarts Bay, respectively 16 km south and 48 km north of the Salmon River.

Mitchell (1981) estimated annual productivity in 1978 in the upriver control Area 50 at 1221 g/m^2 and in 1979 at 1101 g/m^2. Downriver, in Area 53, annual productivity in 1979 was estimated at 796 g/m^2. These estimates were assumed by Mitchell to be the level of productivity exhibited by the natural high salt marsh that was diked and converted to pasture in 1961. They are also comparable to regional productivity estimates for other high salt marshes (Hutchinson 1987). Mitchell conservatively estimated the productivity of the diked pasture in 1978 at 1200 g/m^2, a value similar to that of high salt marsh.

Two years after dike removal, annual net primary productivity in the restoration site diminished to less than 400 g/m^2 (Mitchell 1981). Mitchell anticipated that productivity would increase as colonizing species established and assumed from the trend of Carex lyngbyei and Salicornia virginica that "...a large part of the study site will be producing 1400-1800 g/m^2/yr."

Table 19 and Figure 33 show trends in annual net primary productivity in the restoration area and productivity levels in the controls. Mitchell's productivity prediction for the restoration area was exceeded by 30-60 percent. Estimates of productivity of 2300 g/m^2/yr are higher than most salt marshes in the region, an observation that we attribute to stored nutrients and rapidly establishing communities in areas with little competition. We anticipate that present levels of productivity in the restoration area will be sustained for a number of years and gradually diminish to levels more characteristic of the region on the order of 20-40 percent less than in 1988.

Table 19. Trends in estimates of above ground net primary productivity in g/m^2/yr in the restoration area and controls from 1978 to 1988 in the Salmon River estuary north shore study site.

Site/Area	Year				
	1978	1979	1980	1984	1988
Control Area 50	1221	1101			
(n)	(11)	(10)			
Control Area 50		796			
(n)		(30)			
Restoration Area 51 & 52	1200*		296	1185	2267
(n)	(?)		(49)	(49)	(47)

* estimated by multiplying standing crop peak biomass by 1.5. The same factor was used for two minor species (AGAL and HOBR) in other calculations for which seasonal species biomass curves were unavailable.

Productivity estimates differed by plant community. Table 20 shows productivity trends from 1980 to 1988 for three aggregated community groups. All three groups show similar levels of productivity increase from values of less than 1000 g/m^2/yr in 1980 to levels greater than 1700 g/m^2/yr. The Potentilla pacifica community group consisted of mostly residual species (Potentilla, Agrostis alba, and Juncus balticus) but these increased in productivity considerably as did the colonizers characteristic of the other two community groups. Of course, we have seen that the Carex group dominates the restoration area and accounts for the high productivity for the entire restoration area.

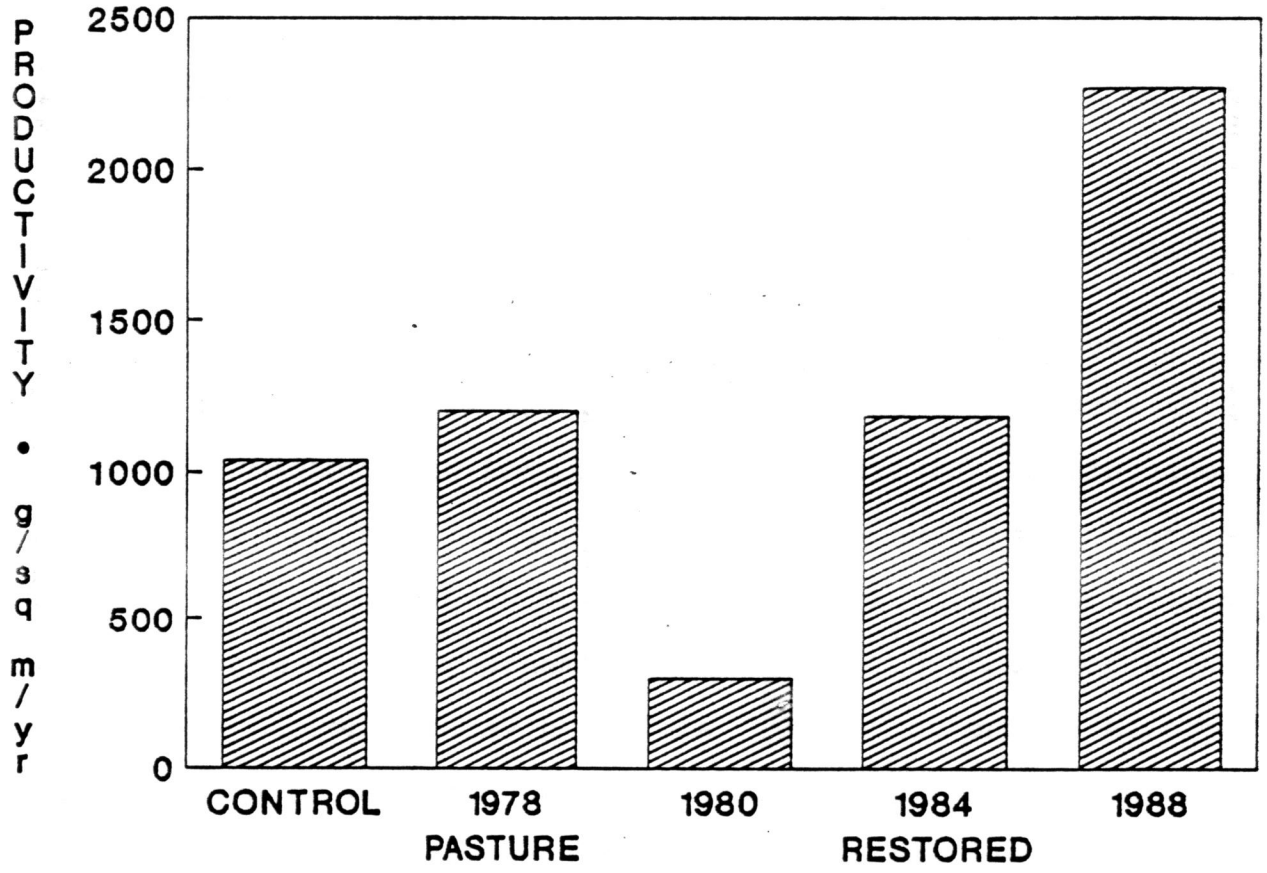


Figure 33. Estimated above ground net primary productivity trends in the restored area of the Salmon River estuary north shore study site from 1978 to 1988 and in the control (average net primary productivity for Area 50 in 1978 and 1979 and Area 53 in 1979).

Table 20. Trends in estimates of above ground net primary productivity in g/m^2/yr for three plant community groups in the restoration area and controls from 1978 to 1988 in the Salmon River estuary north shore study site.

Community Group	Year		
	1980	1984	1988
Carex lyngbyei	224	1080	2277
(n)	(29)	(29)	(29)
Salicornia virginica	67	619	1677
(n)	(8)	(7)	(8)
Potentilla pacifica	1030	1384	2386
(n)	(6)	(5)	(6)

Diversity

Vascular plant and community diversity are ecosystem properties which have changed over the eleven years of restoration. Diversity can be analyzed with respect to (1) a richness component (i.e., species number), and (2) an evenness component (i.e., regarding the relative proportions of species) (3) a dominance component (i.e., emphasizing the most important species in the community), and (4) diversity across an ecological gradient. Two kinds of gradient were observed in the restoration area, an elevation gradient and a temporal gradient. Measures of diversity components reported here are for **richness**, the mean number of species per 0.1 m^2 plot and a statistical estimate of number of species in a community given an unlimited plot size or number of samples; for **evenness**, the Shannon-Weaver index; and for **dominance**, the inverse of the Simpson index (Magurran 1988).

Plant species diversity increased in richness and evenness, and diminished in dominance, over the elevation gradient (Table 21). C. lyngbyei, the lowest elevation community, had the fewest species (ca. 9 species in and infinitely large community) and exhibited high dominance (low inverse Simpson index). The high salt marsh with J. balticus, on the other hand, had the highest species number (ca. 19 species in an infinitely large community) and the lowest dominance indicated by inverse Simpson index values close to 5. Confounding the role of elevation in explaining gradient diversity, is that the highest community is a low diversity **freshwater** wetland (Oenanthe sarmentosa-Scirpus maritimus community.

Although Mitchell's and our data is amenable to a temporal analysis of diversity, we have not had suffient time to complete our analysis. The general trend, however, was a sudden diminishing of richness when the dike was removed and saline water intruded. This action killed the upland species. During the early stages of colonization, richness increased as ephemeral colonizers increased. Later, with the consolidation of the low salt marsh communities, richness again decreased with the competitive loss of many of the ephemeral colonizing species. It will take many years for richness diversity component to recover.

Evenness and dominance component trends, as measured by the Shannon-Weaver and Simpson indicies, are more complex than the richness trends as the initial stages of colonization (1979-82) were marked by low dominance. This relatively equable situation changed to high dominance communities starting in 1984.

Table 21 . Selected diversity measures for eight plant communities in the north shore study site of the Salmon River estuary in 1988 based on 450, 0.1 m plots. n = number of samples, standard error of the estimate in parentheses. Elevation in m NGVD.

	Community							
	CALY	CAPO	AGPO	POJU	JUDE	OEPO	OESC	SADI
n	171	40	50	32	40	12	12	93
Mean Elev.	1.184	1.359		1.472		1.651		1.267
Avg. No. Spp/Plot [N]	1.3	2.9	3.7	4.4	4.9	4.7	2.5	3.7
Est. No. Species/ Commun. [N0]	9 (0.00)	8 (0.98)	17.9 (1.66)	18.9 (1.35)	19.9 (2.16)	15.7 (1.44)	11.7 (2.07)	13.0 (0.00)
Shannon-Weaver [N1]	1.2 (0.05)	2.0 (0.16)	4.3 (0.29)	5.0 (0.64)	6.3 (0.40)	5.1 (0.25)	3.1 (0.35)	4.9 (0.29)
Inverse Simpson [N2]	1.1 (0.02)	1.5 (0.10)	3.3 (0.29)	3.4 (0.37)	4.8 (0.26)	4.7 (0.22)	2.7 (0.30)	3.5 (0.22)

Ecological Assessment of South Shore Study Site

Vegetation

Pre-dike removal vegetation. We conducted a reconnaissance floristic survey in the 63 ha pasture prior to dike removal in 1987. The 1981 and 1984 air photos showed a distinct vegetation line in the area east of Rowdy Creek and west of the eastern dike (see Figure 6), but we detected no significant vegetation change across this line. Four pasture communities are evident (Table 22 and Figure 34). A few small patches of freshwater marsh dominated by Conioselinum pacificum or Oenanthe sarmentosa often with Typha latifolia or Scirpus microcarpus were scattered at the upland edge of the pasture (Community A in Table 22). Much of the pasture was wet and was dominated by a Potentilla pacifica-Holcus lanatus-Conioselinum pacificum community (Community B in Table 22). A slightly more degraded pasture, mostly to the east of Rowdy Creek, was dominated by a P. pacifica-H. lanatus-Juncus effusus community (Community C in Table 22). Because of a faulty tide gate, a brackish high salt marsh community prevailed along margins of Rowdy creek (Community D in Table 22). These communities differed somewhat from the pasture types described by Mitchell (1981), but generally were similar in having prevalence of both P. pacifica and H. lanatus. The large number of species in the south shore pasture is probably related to its larger size and the fact that there were areas of both brackish and freshwater in different parts of the pasture.

Table 22. Association table for pre-dike removal pasture communities in the Salmon River estuary south shore restoration site based on a 1987 reconnaissance survey. Cover classes: 2 < 10 %; 4 = 10-37 %; 5 > 38 %.

Species	Plots			
Community Group	A	B	C	D
Plot Number	444 967	22333333333444440111222222223460 126012346789012452589013457895836	00011111000010 712464589017	0141 1733
Hypochaeris radicata	---	----22-22-22-2--22----------	----------	----
Lolium perenne	---	2--2---2---2-22222----22----	-----2----	----
Barbarea orthoceras	---	222-22222222222--2--2222--22-	----------	----
Plantago lanceolata	---	2-------2222-----2---22------	----------	----
Pteridium aquilinum	-22	--2----2----------------------	----------	----
Digitalis purpurea	---	--22--2--22---22-------------	----------	----
Epilobium watsonii	---	--2-22224224222-------------	----------	----
Vicia sativa	---	--2---2-2----2--------------	----------	----
Scirpus microcarpus	-5-	----------------------------	----------	----
Typha latifolia	--5	----------------------------	----------	----
Trifolium wormskjoldii	--2	2-222--2--22---222222222----	----22-----2	-2-
Conioselinum pacificum	555	--2242422222455-222--2--2-245	222-22----2	---
Achillea millefolium	---	2-22222--224--222422--22-22222	2------2-2-	----
Cirsium arvensis	---	2222-2-22222222------2-2-----	2------2---	----
Anthroxanthum odoratum	---	2-------------2-2---2--2----2	2----------	----
Rumex acetosella	---	---2422-54242222---22--22--2--	2----22----	----
Poa pratensis	---	----------2--222-222-------	-2---2-----	----
Potentilla pacifica	-22	555555554455545525555555555555	52-555522242	4254
Holcus lanatus	-22	55555555555555552-455425222225	54545455-554	----
Rumex occidentalis	---	22-22222-22222222-222222222222	222222222-22	--2-
Rumex crispus	---	2--2---22--2--2--2----2-22----	22-222--2-2-	----
Deschampsia caespitosa	---	------2----------4---24--22-4-	2------2--2-	2-2-
Aster subspicatus	---	-----2-----22-2-5-2---2422222-	4--424--2--2	----
Trifolium repens	---	--222----2--------2---------2-	-----2-2-2--	----
Festuca arundinacea	---	2------2---------2----------	--2--2------	----
Lotus corniculatus	---	5424422--4------554222222-----	542244452442	----
Rumex discolor	---	--22-----2-2----------22--	2---2-22----2	----
Carex lyngbyei	---	----------------------2-2--	----------2	5-22
Juncus balticus	---	-------2-2-----2-----22-222-2-	22222-22222	4452
Agrostis alba	-2-	----------244422422--2-2	4555555525555	5-44
Cotula coronopifolia	---	----------2----------	----------	22--
Stellaria calycantha	---	----------22-2------	2----22----2	----
Juncus effusus pacificus	---	----------2-22-22-------	-2---2242-2	----
Hordeum brachyantherum	---	----------22----------	----222----	----
Distichlis spicata	---	----------------------------	----------	5-5-
Atriplex patula	---	----------2----------	----2------	224-
Scirpus maritimus	---	----------------------------	----------	252-
Galium triflorum	---	----------2-2------	222-222-222	2--2
Angelica lucida	---	----------2----------2	2224---2-2-2	----
Juncus effusus gracilis	---	----------25454--24455	---5	
Alepecurus geniculatus	---	----------22----42222	----	
Eleocharis palustris	---	----------25222	-425	
Polygonum spp.	---	----------2--2----2	---2	

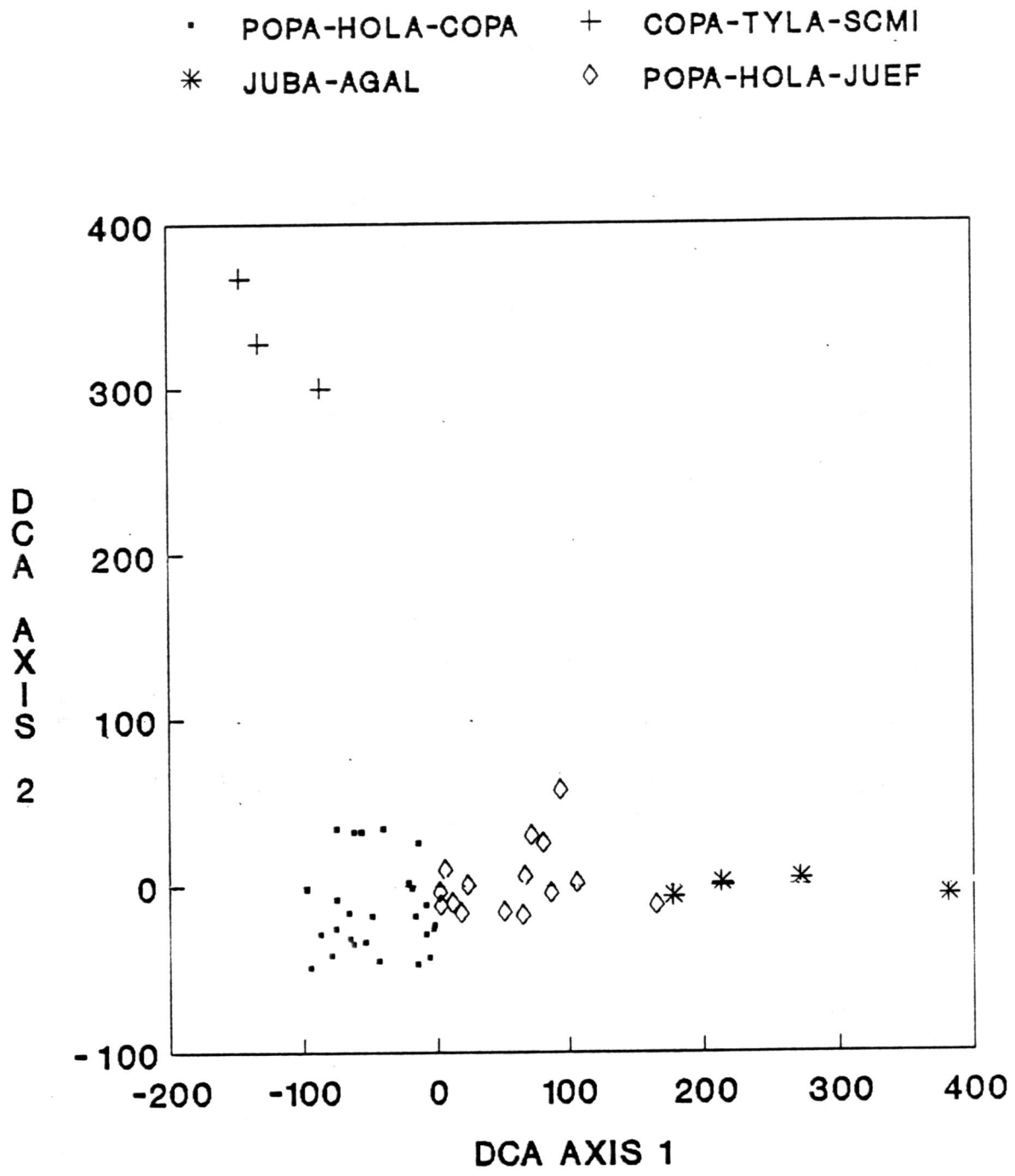


Figure 34. Detrended correspondence analysis of reconnaissance floristic data from the pasture prior to dike removal in 1987 in the south shore study site, Salmon River estuary; POPA-HOLA-COPA = Potentilla pacifica-Holcus lanatus-Conioselinum pacificum comm.; JUBA-AGAL = Juncus balticus-Agrostis alba comm.; COPA-TYLA-SCMI = C. pacificum-Typha latifolia-Scirpus microcarpus comm.; POPA-HOLA-JUEF = P. pacifica-H. lanatus-J. effusus comm.

The ordination (Figure 34) demonstrates the floristic distinctiveness of the freshwater and brackish communities and also a generalized moisture/salinity gradient along DCA Axis 1.

We also surveyed vegetation along Transect C in Area 56 and in scattered localities in Area 54 on the sides of the restoration area (see Figures 3 and 6). This sampling of high marsh served as a "control" on the south shore restoration similar to the arrangement of controls on the north shore.

Post-dike removal vegetation. As with the restoration area in the north shore study site, most of the pasture vegetation on the south shore was killed by saline water intrusion. In 1988, almost 80 % of the restoration was bare of live vascular plant cover. There was much cover by an algal ooze in shallow depressions and elsewhere by dead pasture vegetation, still rooted in place, but matted. The most prominent vascular plant was Potentilla pacifica, residual from the pasture, but showing stress except along creek margins. Although we collected floristic data from the permanent plots along transects in 1988, the year after dike breaching, we did not analyze the data for community characterization. Instead, we report summary frequency data collected from 688 nested plots to be analyzed by the nested frequency method upon future resurveys.

Table 23. summarizes nested percent frequency of 29 species for three plot sizes (0.01, 0.25, 1.00 m^2) stratified into five vegetation groups. **Group 1**, the most extensive vegetation type, was characterized by **stressed** P. pacifica, with between 10 and 75% cover. Four species were common in this group, the residuals

Table 23. Percent frequency in three sizes of nested plots for five vegetation groups in the restoration area of the south shore study site, Salmon River estuary. Species acronyms are given in Appendix 2.

Veg. Group[a]	Group 1 (n=268)			Group 2 (n=219)			Group 3 (n=77)			GROUP 4 (N=71)			GROUP 5 (N=53)		
Plot Size[b]	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
POPA	89.6	94.8	95.5	32.0	51.6	56.6	97.4	100.0	100.0	90.1	98.6	100.0	1.9	3.8	5.7
ATPA	17.9	43.7	59.3	3.7	9.6	13.2	15.6	32.5	44.2	18.3	39.4	49.3	0.0	5.7	11.3
SAVI	11.9	29.5	38.4	3.7	13.7	18.7	0.0	3.9	6.5	1.4	5.6	7.0	1.9	7.5	17.0
AGAL	9.3	18.7	24.6	5.0	8.2	10.0	11.7	24.7	28.6	11.3	18.3	22.5	7.5	26.4	30.2
ASSU	1.1	1.9	3.4	0.0	0.0	0.0	1.3	3.9	5.2	2.8	4.2	4.2	0.0	0.0	0.0
HOBR	4.5	10.1	14.9	0.9	0.9	1.4	0.0	3.9	7.8	23.9	39.4	50.7	0.0	1.9	3.8
SPMC	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	0.0	0.0	0.0
SPMA	0.7	1.5	1.9	0.0	0.5	0.5	0.0	0.0	0.0	0.0	0.0	0.0	9.4	26.4	32.1
PLMA	0.0	1.9	1.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
DECE	2.2	4.1	6.7	0.0	0.5	0.9	3.9	7.8	7.8	4.2	9.9	9.9	1.9	3.8	7.5
DISP	0.0	0.4	0.7	0.5	1.4	1.4	0.0	0.0	1.3	0.0	0.0	0.0	0.0	0.0	3.8
JUBA	2.2	3.4	3.7	0.9	2.7	3.2	1.3	1.3	2.6	85.9	93.0	95.8	1.9	1.9	1.9
CALY	1.1	2.6	3.0	0.5	1.4	1.4	2.6	3.9	3.9	0.0	0.0	0.0	1.9	13.2	17.0
COCO	0.0	0.0	0.4	0.0	0.5	1.4	0.0	0.0	0.0	0.0	2.8	2.8	3.8	26.4	43.4
TRMA	0.7	1.1	1.5	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	9.4	24.5	37.7
SCCE	0.7	0.7	1.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
RUOC	0.0	0.4	0.4	0.0	0.0	0.0	0.0	0.0	1.3	1.4	1.4	1.4	18.9	35.8	47.2
CAOB	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	1.4	0.0	0.0	0.0
ANLU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0
ACMI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	1.4	0.0	0.0	0.0
HYRA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0
PUPU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	0.0	0.0	0.0
SOAR	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	1.4	0.0	0.0	0.0
GATR	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.4	1.4	1.4	0.0	0.0	0.0
TRWO	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	5.6	7.0	0.0	0.0	0.0
HOLA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	2.8	0.0	0.0	0.0
FERU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8	4.2	5.6	0.0	0.0	0.0
GLMA	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.9	3.8	7.5
JUBU	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.8	5.7	5.7

Table 23. Continued

[a]Vegetation Groups: 1 = stressed; POPA and/or ATPA cover 10-75 %
2 = bare; POPA and ATPA cover <10 %
3 = vigorous; POPA and/or ATPA cover >75 %
4 = appreciable JUBA cover >10 %
5 = appreciable COCO cover >10% (removed dike)

[b]Nested Plot size: 1 = 10x10 cm; 2 = 50x50 cm; 3 = 100x100 cm

P. pacifica and A. alba, and the colonizers A. patula and S. virginica. **Group 2**, also widespread, was essentially **bare** and had the same species although their cover was very minor (<10 %). **Group 3**, typically occupied slightly higher ground and lined creek margins. It was characterized by **vigorous** growth of P. pacifica. Generally S. virginica was absent from this group. **Group 4**, marked by the presence, but often not heavy cover, of J. balticus was commonly associated with P. pacifica, A. patula, and Hordeum brachyantherum. Finally, **Group 5**, characterized by Cotula coronopifolia cover of >10 %, was found in areas heavily trampled by cattle and horses in the upper part of the wetland and along the former east dike (Figure 6). Associated in this group were Carex and Triglochin suggesting not only disturbance but also lower elevation related to subsidence of the fill material that was placed into the former ditch which paralleled the Rowdy Creek dike. When the area was diked to make pasture, this ditch served as conduit for Rowdy Creek flow to the Salmon River. The ditch eroded deeply because of Rowdy Creek flow and twice daily tidal scouring. There was insufficient fill material to compensate for the increased depth of this creek.

Our analysis of the south shore restoration area one year after dike breaching shows greater bare ground cover than occurred on the north shore one year after breaching. It shows the same mortality of pasture species and the persistence of residuals, Potentilla and Agrostis, but the latter was quite minor. We account for this difference in bare ground and algal mat by the observation of the south shore restoration area being 19 cm **lower** than the north shore restoration area at time of dike removal (mean elevation of 0.96 vs 1.15 m NGVD) as shown in Figure 10. The implications of this lower surface elevation at 0.91 MLLW (= 0.96 NGVD) which is **lower** than the mudflat colonization zone commonly found in Oregon salt marshes (Eilers 1975, Frenkel et al. 1978) is difficult to assess. We suspect that recolonization and restoration of the south shore area will be slower than has been the case on the north shore.

Elevation

We have already presented the observation of subsidence of the south shore restoration area where there was a mean subsidence of 36.7 cm from the level of former high marsh over a period of 27 years (1961-1987).

Elevation measurements were taken at five to seven spots within one meter of each of 49 permanent plots along 5 transects distributed throughout the restoration area (Figure 6). Mean elevation of the restoration area was 0.959 m NGVD (s.d. 0.075) (s.e. 0.001). Elevation ranged from 0.718 to 1.187 m NGVD. Multiple elevation measurements at each permanent plot permitted

assessment of surface roughness. The mean standard deviation of 49 plots was 0.0287 m NGVD (range 0.004-0.096). In the field we observed differences in roughness in different sections of the restoration area, but we were unable to relate these differences to any particular factor.

DISCUSSION

Composition

The developing communities in the north shore restoration area are typical in composition to many relatively young Pacific Northwest salt marshes (Burg et al. 1980, Eilers 1975, 1976, Frenkel et al. 1978, Frenkel and Boss 1988, Jefferson 1975, Macdonald 1977, Seliskar and Gallagher 1983). The relative zonal arrangement of these communities, although poorly expressed, is also in accord with that described by Eilers (1975), Frenkel et al. (1978), Jefferson (1975), and Seliskar and Gallagher (1983). A nearly monotypic Carex lyngbyei community extended over much of the upriver two-thirds of the restored marsh (Figure 8), in places almost abutting the upland. In this regard, the spatial pattern in the restored site differed from other Pacific Northwest salt marshes where there was usually a strong zonal grading with extensive areas of high and intermediate elevation marsh. The high communities in the restored area were relegated to a very narrow fringe adjacent to the upland. Restored area communities had little similarity with those in the upstream and downstream control areas.

Mitchell (1981) reported almost complete mortality of upland pasture species in the north shore study site upon the intrusion of saline water. We observed this same mortality in the 63 ha south shore site following the 1987 dike removal. These observations are in accord with other observations in the Pacific Northwest (Dawe and McIntosh 1981). Residual high salt marsh

species Potentilla pacifica and Agrostis alba persisted, but eleven years after restoration, diminished in cover. Two groups of colonizers appeared in the restored area in the first through third year after breaching, ephemeral colonizers and permanent colonizers. Ephemerals (e.g., Cotula coronopifolia, Puccinelia pumila, and Spergularia maritima) rapidly occupied lower elevation mud flat and persisted a few years before disappearing. Many of these were annuals. Permanent colonizers (e.g., C. lyngbyei, Distichlis spicata, Juncus balticus, and Salicornia virginica), were slow to establish, but by the fourth year after restoration became more prominent. Presently, they dominate the restored area. Dawe (personal communication 9 February, 1990) reported a similar pattern for a marsh restoration in the Englishman River estuary in British Columbia, except for C. lyngbyei, which in the fourth year showed dieback, and ten years after restoration comprises a very minor component of this British Columbia restored marsh. We have not observed such a dieback in the Salmon River, and in 1989 Carex was still the dominant in the restored area.

Structure

Elevation

By affecting frequency and duration of tidal water, surface elevation exerts a primary control on salt marsh composition and processes. Since the north shore restoration site apparently subsided 30-40 cm over the 17 years that it was diked (Mitchell 1981), upon dike removal, this lowered elevation led to increased flooding, establishment of low marsh species, and development of

low marsh communities. In 1988, the mean elevation difference between the control and restored areas was 34.9 cm.

This elevation difference must be viewed within a complex framework involving a number of factors. First, from 1961 to 1978 when the dike was breached, the diked pasture subsided. Mitchell (1981) attributed this subsidence to compaction and loss of soil buoyancy. Her measurements of bulk density and soil organic matter in the control and pasture suggested that oxidation was an unlikely cause of subsidence. Second, since the flanking natural controls were not subject to grazing but were affected by sedimentation associated with tidal flooding from 1961 to 1978, we anticipated an increase in elevation during the time that the ajacent diked wetland was subsiding. Third, during the 1978 to 1988 restoration period, elevation of both the control and restored areas increased.

Consider first, the relative elevations in 1978. We observed an elevation increase in the control areas of 2 to 5 cm over the restoration period (1978-1988). Jefferson (1975) reported "accretion" rates of 0.5 to 1.7 cm/yr in low silty Oregon marshes. These observations suggest that over the 17 years prior to 1978 (the time period that the pasture was diked), the controls may have increased in elevation from 3 to 8 cm. The net effect of this increase in elevation of the control and subsidence of the diked area in 1978, was the observed 30 to 40 cm difference between control and pasture elevation.

Regarding the elevation status of the north shore study site in 1988, elevations increased in both the controls and restored area from 1978 to 1988. The controls increased from 2 to 5 cm compared with the restored area which increased from 3 to 8 cm. It would appear that the restored area is increasing in elevation at a greater rate than the controls, but future measurements are required to determine whether this differential will continue. It may be that as the elevation of the restored area approaches that of the control, the rate of increase in the restored area will diminish.

Elevation changes were not uniform over the restored area. Low areas increased in elevation more than higher areas (Figure 14). Elevation increases were greater in areas of high tidal exchange than low tidal exchange. We conclude that tidally contributed sediment accounts for this differential growth in the restored marsh surface.

Species and plant communities in the Salmon River restored area are established at somewhat lower elevations than reported elsewhere in the literature. For example, Eilers (1975, 1976) observed monotypic stands of Carex lyngbyei on West Island, Nehalem estuary between 1.00 to 2.10 m MLLW compared with the Salmon River restored area range of 0.95 to 1.30 m MLLW. In Joe Ney Slough in Coos Bay, Carex ranged from 1.70 to 2.00 m MLLW (Taylor 1983). The same is true of the Salicornia-Distichlis community which in the restored area ranges from 1.00 to 1.40 m MLLW compared with 1.50-1.70 m MLLW in Joe Ney Slough (Taylor

1983). The elevational basis for these comparisons in MLLW and the differences could have been because of an inaccurate calculation of MLLW (Mitchell 1981) or because of estuary to estuary differences in the establishment of species; the latter may relate to the relative position of the marsh with respect to the estuary.

High marsh species and communities were established as narrow bands on flattened dikes. Elevation ranges of species and communities were also somewhat lower (ca. 50 cm) on flattened dikes than in natural salt marshes reported in the literature (Eilers 1975, 1976, Frenkel and Boss 1988, Taylor 1983).

Creeks

Tidal creeks deepened and narrowed upon tidal reconnection. In the lower to middle reaches, relative deepening was about 40-60 cm; in the upper reaches this diminished to 5-10 cm. Increased creek depth facilitates marsh drainage and can affect vegetation patterns. We did not observe this relationship, however.

Soils

Over an eleven year period, about 2-6 cm of sediments accreted in the restored area above the 1978 pasture surface. Low elevation areas tended to accrete more than high, supporting the explanation that sediments were tidally deposited. Sediments were sandier in the downstream portion of the restoration area than upstream. Organic content of the soil tended to increase toward the upland where tidal inundation and exchange was less

and where decomposed litter accumulated locally and was not flushed to the estuary. Organic matter was least in the area of heavy tidal exchange. Salinity was higher in the downstream, sandier area than upstream.

Plant community distribution accorded closely to soil characteristics. Sandier and more saline areas supported a Salicornia-Distichlis plant community. Fine silt and more brackish soils supported the monotypic Carex community which accorded more closely to the tall form of Carex than to the short form (Eilers, 1975, 1976, Frenkel et al. 1978, Frenkel and Boss 1988, Jefferson 1975, Macdonald 1976, Seliskar and Gallagher 1983, Taylor 1983).

Function

Two aspects of function, productivity and diversity, were analyzed. The relatively intact salt marsh flanking the restoration area had approximately the same level of biomass and productivity as the pasture in 1978-1979 (about 600 g/m^2 and 1200 $g/m^2/yr$ respectively). Upon dike removal, biomass and productivity [estimated by the Kibby et al. (1980) method] diminished greatly with mortality of upland species. Recovery of biomass and productivity was slow until five to six years after breaching. Eleven years after dike removal, biomass was 2.7 times that prior to breaching (an increase from 600 to 1645 g/m^2). Likewise annual above ground net primary productivity increased from about 1200 g/m^2 to almost 2300 g/m^2. This level of productivity is comparable to the highest values reported for

Pacific Northwest salt marshes (Eilers 1975, 1979, Hutchinson 1986, Macdonald 1984). We interpret the enhanced productivity to relate to the lower elevation of the marsh surface, presence of released nutrients from decomposed pasture material, and relatively high tidal exchange of nutrients (Steever 1976). We anticipate that elevated productivities will diminish in the future.

Generally, plant species diversity diminished over the restoration period. Diversity was heavily influenced by development history, dropping precipitously with tidal reconnection, increasing during the initial colonization period, and later decreasing with consolidation of the restored marsh surface by a few dominant plant communities. We regard diversity as a poor indicator of marsh function, one which is very difficult to measure and interpret.