Floristic Development Patterns in a Restored Elk River Estuarine Marsh, Grays Harbor, Washington

Ronald M. Thom[1,3]
Robert Zeigler[2]
Amy B. Borde[1]

Abstract

We describe the changes in the floral assemblage in a salt marsh after reconnection to estuarine tidal inundation. The Elk River marsh in Grays Harbor, Washington was opened to tidal flushing in 1987 after being diked for approximately 70 years. The freshwater pasture assemblage dominated by *Phalarais arundinacea* (reed canary grass) converted to low salt marsh vegetation within 5 years, with the major flux in species occurring between years 1 and 4. The system continued to develop through the 11-year post-breach monitoring period, although change after year 6 was slower than in previous years. The assemblage resembles a low salt marsh community dominated *by Distichlis spicata* (salt grass) and *Salicornia virginica* (pickleweed). Because of subsidence of the system during the period of breaching, the restored system remains substantially different from the *Deschamsia cespitosa* (tufted hairgrass)-dominated reference marsh. Use of a similarity index to compare between years and also between reference and restored marshes in the same year revealed that similarity in floral composition between year 0 and subsequent years decreased with time. However, there was a period of dramatic dissimilarity during years 1 to 3 when the system was rapidly changing from a freshwater to estuarine condition. Similarity values between the reference and restored system generally increased with time. Somewhat surprisingly the reference marsh showed considerable between-year variation in similarity, which indicated substantial year-to-year variability in species composition. Based on accretion rate data from previous studies we predict that full recovery of the system would take between 75 and 150 years.

Key words: dike breach, Grays Harbor estuary, Pacific Northwest wetlands, salt marsh restoration.

Introduction

We describe temporal changes in the salt marsh floral assemblage after reconnection, by breaching the surrounding dike, to estuarine tidal inundation. We apply the results of 11 years of monitoring to address issues such as successional vegetation patterns, time frame for development, use of reference sites to assess performance of the system, and monitoring parameter selection—issues relevant to understanding the effects of dike breaching on restoration of estuarine marshes.

Tidal marshes were once extensive in Washington State, but by the mid-1800s dikes had been constructed around vast expanses of tidal marshes to promote the use of these areas for cattle grazing (Nesbit 1885; Thom & Hallum 1991). The practice of diking tidal wetlands continued well into the 20th century. In Puget Sound alone over 70% of the tidal wetlands were lost to diking or development (filling and dredging) by the mid-1950s (Thom & Hallum 1991). Tidal marshes occupy the lowlands in an otherwise steep terrain formed by the Cascade Mountain Range to the east of Puget Sound and the Olympic Mountains between Puget Sound and the Pacific Ocean. Major earthquakes every 300 to 700 years resulted in immediate subsidence of coastal regions on the order of 1 m, and subsequent tsunamis buried coastal marshes with sand (Atwater 1987). As a consequence tidal marshes are relatively young compared with marshes of the East and Gulf Coasts. Because they are "wedged" between a steep topography on both the landward and seaward sides and are "reset" by periodic tectonic events, tidal marshes represent an ecosystem with extreme natural pressures. They are generally regarded as essential elements of the estuarine landscape and provide habitat for a variety of species, as well as being sources of primary production for the general estuarine system (Thom 1987). Restoration of tidal marshes is a priority of regulatory agencies and conservation groups in the region (Weinmann & Kunz 1994).

In the Pacific Northwest the U.S. Army Corps of Engineers (Patrick Cagney 2000, personal communication,

[1]Battelle Marine Sciences Laboratory, 1529 W. Sequim Bay Road, Sequim, Washington 98382, U.S.A.
[2]Washington State Department of Fish and Wildlife, 600 Capitol Way N., Olympia, Washington 98501, U.S.A.
[3]Address correspondence to R. M. Thom, ron.thom@pnl.gov.

Seattle District, Corps of Engineers), U.S. Fish and Wildlife Service (Tanner et al. 2002, this issue), the National Oceanic and Atmospheric Administration (Cornu & Sadro 2002, this issue), and other agencies and groups are actively engaged in dike breach projects to restore tidal marshes. Although there is vigorous activity in dike breaching to restore tidal marshes in the Pacific Northwest, there are very few published records on patterns and rates of development of these systems. Furthermore, because many of these projects are carried out to mitigate losses of tidal marsh habitat, there is a paucity of published information to support dike breaching as an effective method for compensating impacts to existing marshes. As is the general case with tidal wetlands (Zedler 2001), we cannot reliably predict rates and patterns of development of the ecological structure and function of restored systems. Comprehensive information on development and ecological functions of restored marshes available in the Pacific Northwest comes from projects in the Salmon River in Oregon (Frenkel & Morlan 1990) and the Gog-Le-Hi-Te wetland in Washington (Simenstad & Thom 1996). In 1978 a dike was breached in the Salmon River estuary that re-exposed 21 ha of former tidal marsh to tidal inundation. Over the first 11 years the vegetation shifted from freshwater pasture grasses to tidal brackish marsh. In 1985 and 1986 the Gog-Le-Hi-Te system was created by excavating fill down to former wetland elevations and reopening the marsh to tidal inundation. Through the first 7 years this system experienced extreme sedimentation, which threatened the system's support of fish resources. As a function of infilling, the vegetation shifted from *Carex lyngbyei* (lyngby sedge) that was planted to *Typha* spp. (cattails), and tidal channel morphology and location were greatly altered. Observations made 13 years after construction showed that the patterns of change appear to have slowed since the first 7 years (Thom et al. 2000).

Description of the System

The Elk River restoration site is located in the south bay portion of Grays Harbor estuary (Figs. 1 & 2). The 23-ha site was obtained by the State of Washington in a mitigation agreement in 1983 to compensate for an approximately 16-ha wetland fill associated with the construction of the Ocean Shores airport. A dike was constructed early in the 1900s to convert the native salt marsh to pastureland. In 1987 the new earthen dikes were constructed along the north and west sides of the site to protect adjacent properties from saltwater intrusion and the portion of the land that contained a 7-ha forested (freshwater) wetland. Before breaching the dike in 1987 the vegetation in the pasture was dominated by freshwater wetland plant species, including *Phalaris arundinacea* (reed canarygrass) and *Juncus effuses* (soft rush), with *Juneus balticus* (baltic rush) and *Potentilla pacifica* (Pacific silverweed) interspersed. Reed canary grass is considered a noxious weed in the region. *Pyrus fusca* (crabapple) and *Picea sitchensis* (spruce) had pioneered, but grazing prevented extensive establishment of woody plants. The dikes in the area tended to be overtopped in extreme tides, creating a salt marsh community immediately landward of the dikes. A single 33-m wide breach was made at the northeast corner of the site in an area where a natural tidal channel existed, which allowed tidal waters to flood the site. The remaining dike, which contains crabapple trees and upland grasses, was left intact except for a small (~0.5-m diameter) culvert located approximately midway along the seaward dike.

Surface water salinity and temperature in the vicinity of the site range from 19 to 30 ppt and 8 to 18°C, respectively (Washington State Department of Ecology, unpublished data for 1997). Tidal range averages 2.3 m. Characteristic of many diked systems, the diked portion of the site had subsided approximately 0.5 to 1.0 m since the dike was installed (unpublished data). Hence, the upper portion of the natural marsh immediately seaward of the dike is at greater elevation than the restored marsh. Tidal waters now flood the site approximately 10 times per month.

Materials and Methods

Study Sites and Plots

We established study sites within the restored marsh and in the natural undiked marsh immediately seaward of the dike (Fig. 1). The natural marsh served as a reference for conditions that likely existed in the restored marsh before dike construction. In 1987, immediately after dike breaching, we established seven species presence/absence 1-m^2 plots that were distributed at 61-m intervals from the breach to the opposite boundary of the marsh (Fig. 1). To further quantify recovery of the site, in 1991 we established a 360-m long transect through the site, along which 1-m^2 plots for species coverage sampling were spaced 10 m apart. In addition, we established 1-m^2 plots at 5-m intervals along a 145-m long transect that spanned the elevation gradient in the reference site. The transects and plots were placed through what visually appeared to be areas containing a floral assemblage representative of each site. The plots were spaced to include all apparent species of plants present along each transect.

Sampling Methods

Sampling of the plots was conducted once during mid to late summer (i.e., July through September), the period of full development of marsh vegetation. The

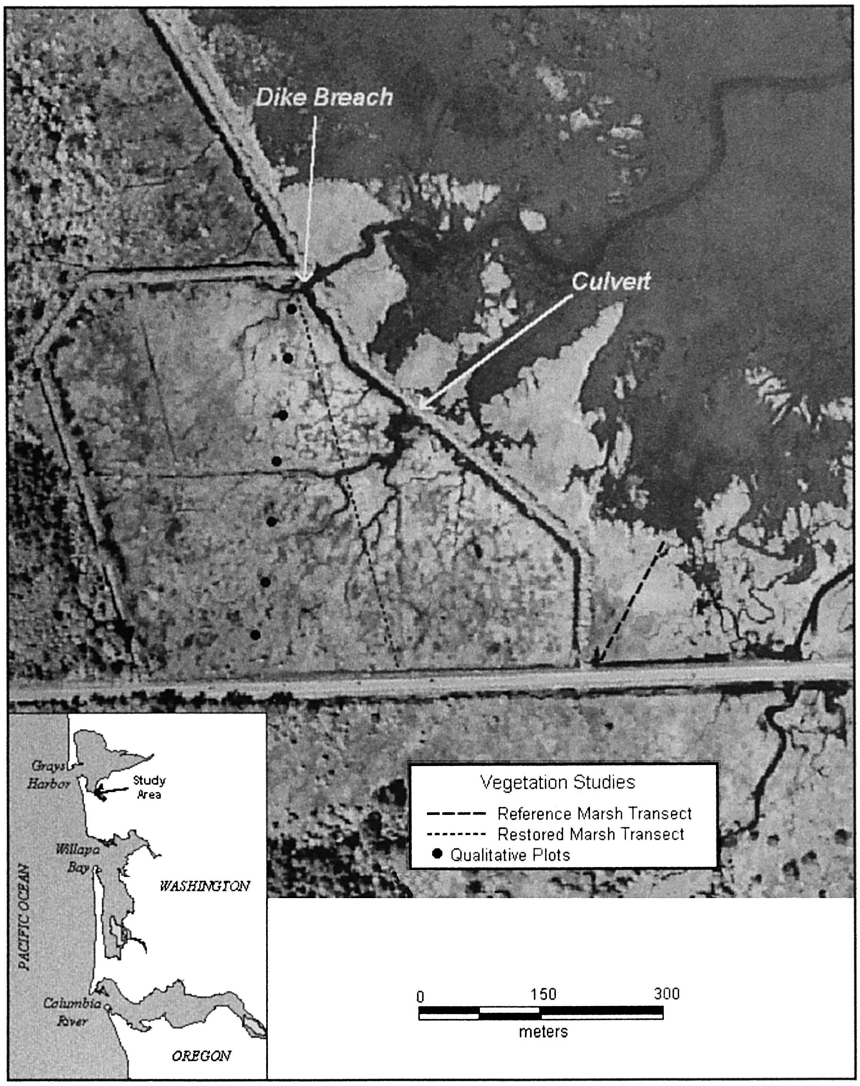


Figure 1. Location of Elk River Marsh sites.

seven initial plots were sampled annually between 1987 and 1998, with the exception of 1995. Only species presence was recorded in these species presence/absence plots. Species were identified and their percentage cover was estimated in the 1-m^2 species coverage plots, which were sampled in 1991 through 1995 and 1998. Percentage cover was estimated visually in 5% increments for each species. Very rare species were given a score of 0.5%. In most years the same person conducted the sampling. In years when more than one person participated in the sampling the person who had been involved in all sampling efforts trained all participants. Calibration exercises among samplers conducted immediately before each sampling showed good correspondence among estimates of cover. Based on this calibration step we believed that variations between estimates made by different individuals were minimized. Anomalous conditions were noted at the plot, such as bare space, evidence of grazing, and disturbances.

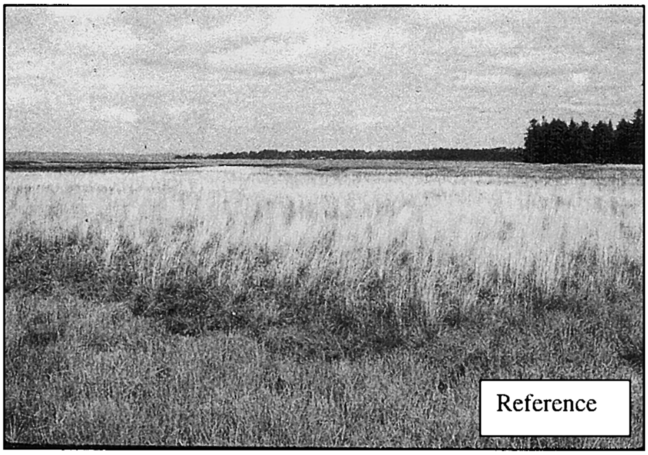
Data Analysis

We used an index (Czekanowski) to estimate similarity in species composition and species cover between years within each site and between sites within each year (Bray & Curtis 1957). The *unweighted* similarity index is calculated as follows:

$$\text{Similarity} = (2a / [2a + b + c]) \times 100\%$$

where a is the number of species in common between two records, b is the number of species exclusive to the first record, and c is the number of species exclusive to the second record.

A *weighted* index of similarity was calculated by including the species and their cover values. For the weighted index, a is the cover values in common by species between the two records, b is the species and their cover values exclusive to the first record, and c is the species and their cover values exclusive to the second record.



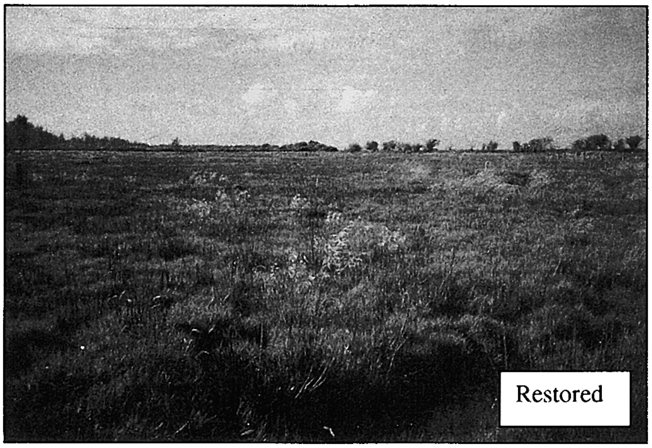


Figure 2. Photographs of the restored marsh (in 1998) and reference marsh. The bright tall vegetation is *Deschampsia cespitosa*, and the darker low growing vegetation is primarily *Distichlis spicata.*

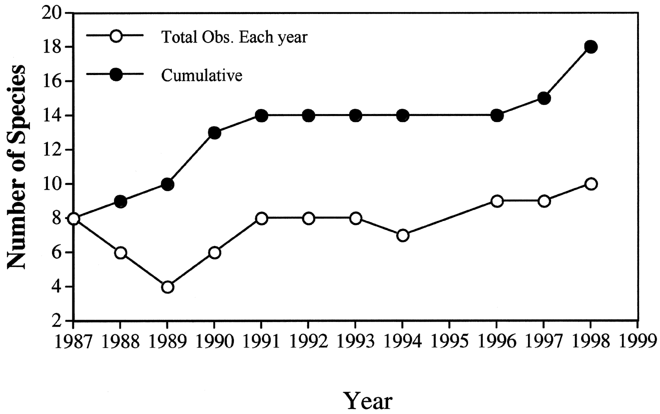


Figure 3. Total number of plant species recorded each year and cumulative number of species recorded through time in the species presence/absence plots located in the restored marsh.

This index ranges from 0.0% (no species in common between records) to 100% (all species and their cover values are the same). The unweighted version calculates similarity in species composition only, whereas the weighted version calculates similarity based on species and cover values. The index has been useful in comparing species composition between sites and years in upland systems (Bray & Curtis 1957), subtidal marine infaunal assemblages (Armstrong et al. 1980), and rocky shore seaweed-dominated assemblages (Prentice & Kain 1976; Thom & Widdowson 1978).

Results

Species Richness

The total number of species noted in the species presence/absence plots within the restored marsh each year varied between 4 and 10, with the lowest number recorded 2 years after dike breaching (Fig. 3). The cumulative number of species (= total number of species recorded at the site since the start of the study) increased for the first 4 years after breaching, leveled off through year 8, and increased dramatically in years 10 and 11. In the species coverage plots the total number of species in the restored site varied between 8 and 12, with no identifiable annual trend (Fig. 4). The reference site contained between 9 and 14 species. The cumulative number of species in the restored site reached 14 by 1994; the same number the reference site contained the year before (Fig. 4).

Plant Assemblage Similarities

The six dominant species (Table 1) provide a strong indication of the assemblage shifts after dike breaching. Based on frequency (= number of plots with a species/seven total plots) of occurrence in the species presence/absence plots, a freshwater *Phalaris*-dominated assemblage shifted to a salt marsh *Distichlis/Salicornia*-dominated assemblage in the restored site 4 to 5 years after dike breaching (Fig. 5). The most rapid shift in species composition occurred during the first 3 to 4 years after dike breaching. *Carex*, a euryhaline taxon, showed the least change through time. *Deschampsia*, the dominant taxon in the reference marsh, was noted in the restored marsh only at the end of the sampling period in year 11.

This shift was verified with data from species coverage plots (Fig. 6A). *Distichlis* cover continued to increase between years 4 and 11, which replaced early enhanced cover of *Salicornia. Triglochin*, often found in places where salt marshes are actively recruiting, increased between years 4 and 7 and decreased between years 7 and 11. *Carex* cover varied little between 1991 and 1998. *Des-*

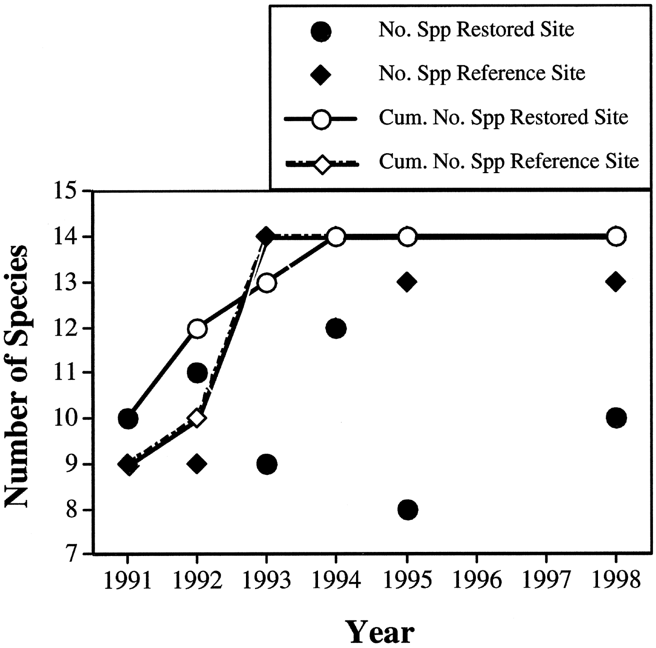


Figure 4. Total number of species and cumulative number of species recorded through time in species coverage plots at the restored and reference marshes.

champsia cover was generally low throughout the period of 1991 through 1998.

Species cover in reference plots showed considerable variation during 1991 to 1998 (Fig. 6B). *Deschampsia* and *Distichlis* showed the greatest range of variability, with *Distichlis* demonstrating a period of dominance between 1993 and 1995. The species observed at the sites changed with time as evidenced by a declining similarity index between 1987 and later records (Fig. 7). The greatest dissimilarity occurred in years 4 and 5 after breaching, when the salt marsh species were increasing in cover most rapidly (Fig. 5).

Although there was a clear trend toward recovery, the reference and restored sites remained dissimilar through year 11 (1998) (Fig. 8). Unweighted similarity showed a general trend upward from a low in year 4 (1991) of 42% and a high in year 11 of 78%. Weighted similarity also increased with time, with the exception of year 11 (Fig. 8). A large increase in cover of *Deschampsia* and a decrease in cover of *Distichlis* at the reference site (Fig. 6B) explain the very low similarity in year 11.

The species presence and coverage at the reference site changed between years as indicated by variation in similarity values calculated between successive years (Table 2). Maximum unweighted and weighed similarities for the reference site were between 70 to 89% and 60 to 84%, respectively. The restored site showed similar between-year variation (Table 2).

In 1998 the restored marsh assemblage showed similarities to the assemblage found at the mid to lower portions of the reference transect. The assemblage along the entire transect in the restored marsh was largely dominated by *Distichlis* and *Salicornia* (Fig. 9A). The reference transect was dominated by *Deschampsia* at the upper end, *Distichlis* in the middle portion, and *Salicornia* the leading (estuary) edge (Fig. 9B).

Discussion

The information on the Elk River site directly addresses a variety of issues regarding dike breaching as a method for restoring tidal marshes in the Pacific Northwest, including successional vegetation patterns, time frame for marsh development, use of reference sites to assess performance of the system, and monitoring parameter selection.

Vegetation Succession Patterns

The succession pattern seen at Elk River is similar to that described for other tide marsh breach systems in the region. The early loss of freshwater vegetation coupled with an increase in bare space (data not presented here), followed by annual and then perennial salt marsh vegetation, resembled the pattern documented by Frenkel and Morlan (1990) for the Salmon River marsh system. The period of most rapid change occurred between years 2 and 5, after which the rate of change in vegetation

Table 1. The six dominant marsh species, their common names, and physical–chemical habitats (Cooke 1997).

Species	*Common Name*	*Salinity*	*Relative Elevation/Hydrology*
Carex lyngbyei	Lyngby sedge	Estuarine to brackish	Mid to low marsh
Deschampsia cespitosa	Tufted hairgrass	Brackish to fresh	High coastal marsh; wet fresh meadows
Distichlis spicata	Seashore salt grass	Marine to estuarine	Mid to low marsh
Phalaris arundinacea	Reed canary grass	Fresh water	Wide tolerance of freshwater flooding
Salicornia virginica	Pickleweed	Marine to estuarine	Low marsh
Triglochin maritima	Seaside arrowgrass	Estuarine to brackish	Low marsh; early mudflat colonizer

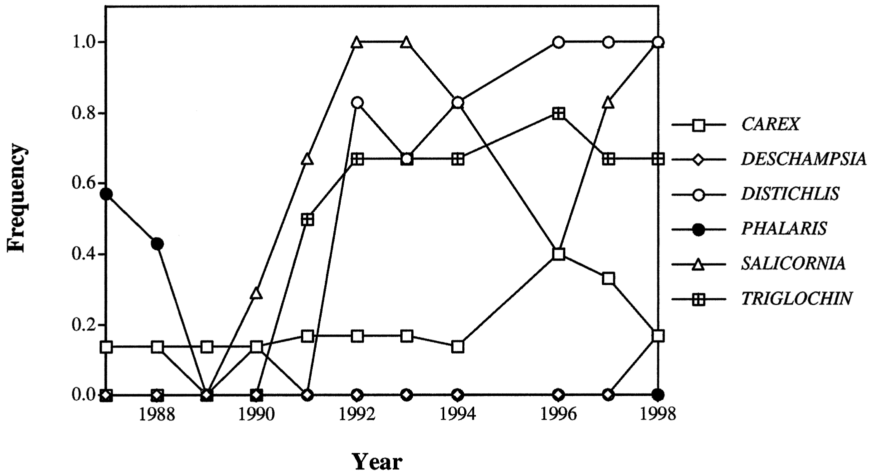


Figure 5. Frequency of occurrence of six major taxa in species presence/absence plots in the restored marsh through time.

slowed. Frenkel and Morlan (1990) noted that ephemeral colonizing species (e.g., *Cotula coronopifolia* [brass buttons], *Atriplex patula* [fat-hen saltbush], *Triglochin maritime* [seaside arrowgrass]) were present during the first and second years after breaching and were replaced by more long-lived species after 6 to 11 years. We observed all these species colonizing bare areas during the first several years of development. At Elk River the noxious invading species reed canary grass was eliminated within 2 to 3 years after dike breaching. It was replaced by species that are common low salt marsh vegetation in the region (e.g., *S. virginica*, *D. spicata*). Reintroduction of tidal action, sediments, seeds, and salty water likely drove these changes. Lyngby sedge showed little change between pre- and post-breach conditions, probably because of its ability to tolerate wide salinity ranges.

The similarity index proved useful in providing a way to summarize multivariate aspects of assemblage change through time. In particular, the index identified that years 2 through 5 were a period of rapid species flux and that the rate of change probably slowed thereafter. The index also indicated that change was not linear through time but exhibited some variability between years. Shorter term studies of dike breaches elsewhere in the region (Snohomish River Delta, Washington, and South Slough of Coos Bay, Oregon) also indicated that rapid species flux occurs within 1 to 3 years

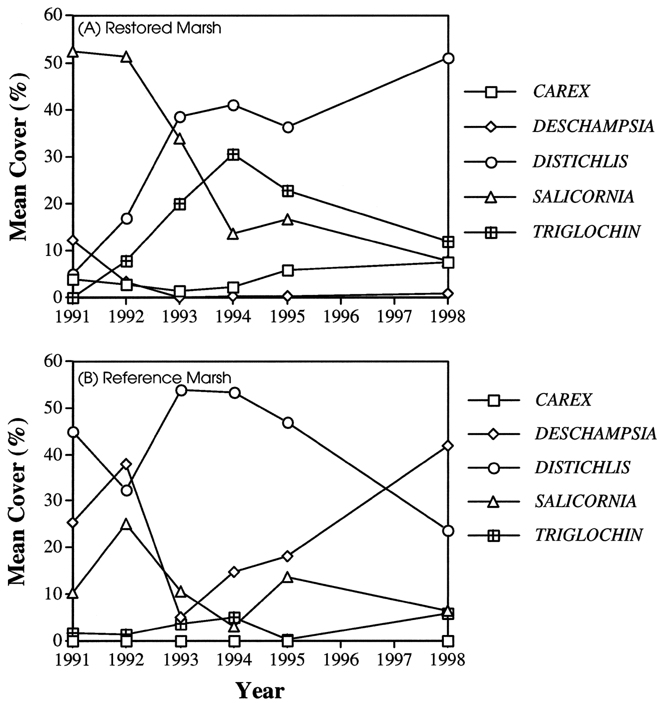


Figure 6. Mean percent cover of five major taxa in species coverage plots in the (A) restored and (B) reference marshes.

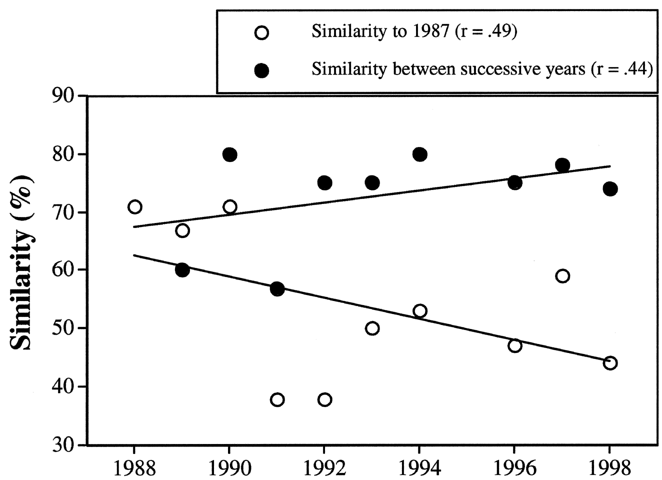


Figure 7. Change in unweighted similarity over time in the restored system. Data from species presence/absence plots. Between successive years means the similarity between the flora in one year compared with the next year.

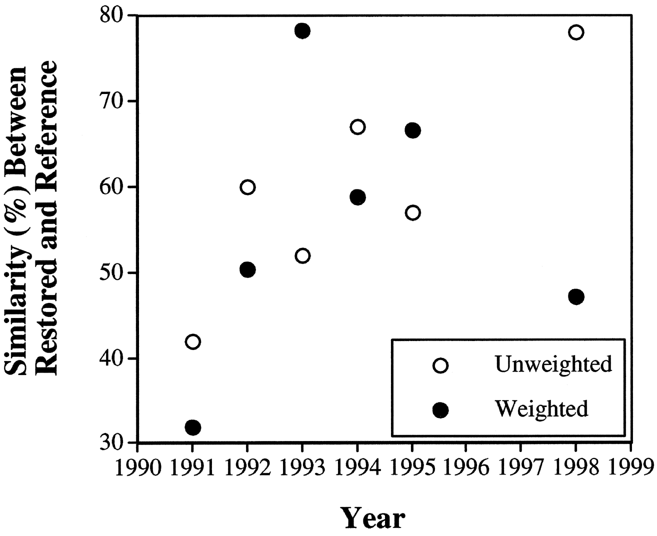


Figure 8. Change in weighted and unweighted similarity over time between the restored and reference marshes. Data from species coverage plots.

after breaching (Tanner et al. 2002, this issue; Cornu & Sadro 2002, this issue).

The restored marsh assemblage resembled the low intertidal portion of the reference marsh. Subsidence on the order of a meter (unpublished data) during the period when the dike was in place accounts for the lower elevation. Based on the natural gradient of habitat types in undiked portions of the area, it was obvious that the marsh within the diked area likely contained mid to high marsh vegetation before diking. Hence, elevation (i.e., hydroperiod) is probably the primary factor controlling the composition of the assemblage (Callaway 2001). Frenkel and Morlan (1990) showed that subsidence-driven elevation declines in their restored system explained why the restored marsh assemblage differed from the nearby reference assemblage. Very limited data on salinity in surface waters in the restored marsh indicate that salinities are high (35–40 ppt) relative to the adjacent estuary waters (19–30 ppt). These hypersalinity conditions are probably caused by evaporation during warm periods (Simenstad & Thom 1996). Because some species now abundant in the reference marsh are limited to fresher conditions, salinity may also be an important controlling factor limiting development of the system to former conditions.

Table 2. Similarity index between successive years at the reference site and restored site.

Years Compared	*Reference Site*		*Restored Site*	
	Unweighted %	*Weighted %*	*Unweighted %*	*Weighted %*
1991–1992	76	74	78	74
1992–1993	70	59	70	66
1993–1994	86	84	92	79
1994–1995	70	79	80	83
1995–1998	89	60	85	78
Average	78	71	81	76

Time Frame for Development

There is uncertainty about the length of time it takes for restored systems to develop. The data from Elk River clearly show that rapid shifts occur within 5 years of dike breaching but continue for a much longer time period, which is similar to rates reported from other systems (e.g., Frenkel & Morlan 1990; Simenstad & Thom 1996; Boumans et al. 2002, this issue; Eertman et al. 2002, this issue). Changes were still taking place at least 11 years after dike breaching. Monitoring programs of 1 to 3 years would be inadequate to capture both periods of rapid change and subsequent slower change.

How long will succession take to reach conditions similar to pre-breach (reference site) conditions? Elevation is probably the main factor controlling vegetation structure within this localized area (Eilers 1975; Cornu & Sadro 2002, this issue; Crooks et al. 2002, this issue). For the restored system to reach the elevation comparable with the restored marsh, approximately 0.5 to 1 m of accretion must occur. In a study of sea level rise and marsh accretion core samples from the Elk River reference marsh showed that annual accretion averaged 6.6 mm/yr for the period between 1963 and 1991 (Thom 1992). At this rate, it would take roughly 75 to 150 years for marsh surface elevation to rise to those comparable with the *Deschampsia*-dominated portion of the reference marsh (Fig. 10). Crooks et al. (2002, this issue) found that the flora in restored sites approximately a century old matched those of reference sites and concluded that vegetation may be restored within a century of dike breaching. Williams & Orr (2002, this issue) showed that accretion in restored tidal marshes in San Francisco Bay was dependent on estuarine sediment supply, erosion of deposited muds, and degree of tidal exchange. Because the restored Elk River marsh has only two relatively small openings to tidal waters and sediment is introduced only through these openings, accretion may be slower than the reference marsh. Development of levee breach marshes to a semblance of reference marsh assemblage in San Francisco Bay, Connecticut, and New Hampshire typically takes 10 to 20 years (Williams & Orr 2002, this issue; Warren et al. 2002, this issue; Morgan & Short 2002, this issue).

Trajectory of Development

Studies cited above are beginning to provide an indication of a pattern of development and recovery of formerly diked estuarine marshes in the Pacific North-

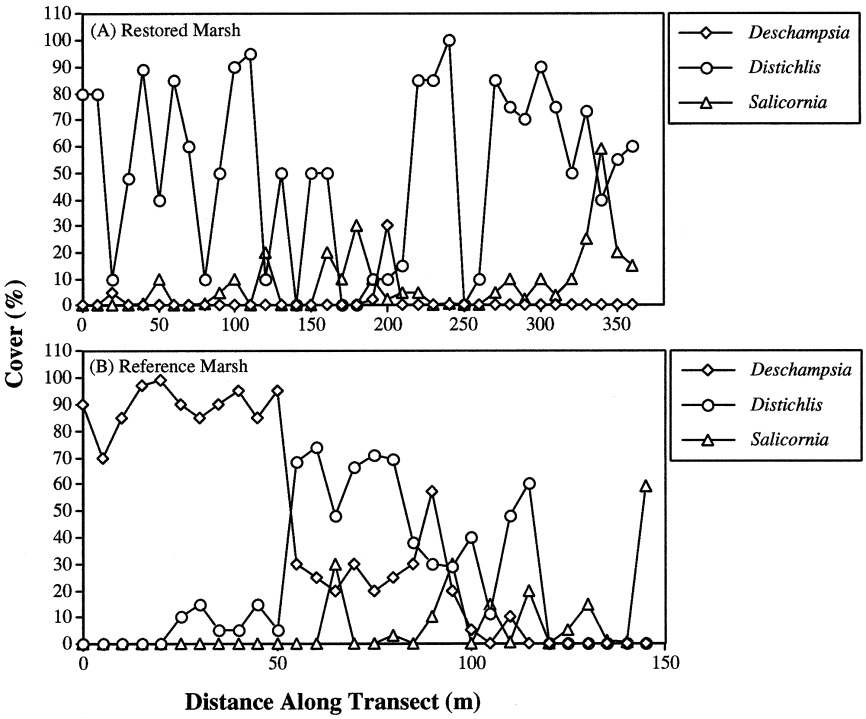


Figure 9. Species cover in 1998 along transects in the (A) restored and (B) reference marshes. Only the three dominant species are shown.

west. There is an initial period of rapid species flux, with a shift to tidal salt-tolerant species by year 5 or 6. Elevation and salinity may affect a slower change after this initial period (Frenkel & Morlan 1990). The similarity index provides an integrated measure of a trajectory of development. The most evident trajectory was the general decline in similarity between the flora in 1987 and subsequent years and the increase in similarity between the restored and reference sites. Using the species presence/absence quadrat data, similarity between the 1987 and the 1998 flora was 44%, indicating an average rate of linear decrease in similarity of about 5%/yr. Based on the species coverage quadrat data the floral assemblages at the restored and references marshes increased in similarity at a rate of about 7%/yr between 1991 and 1998. At this rate the flora in the restored marsh would match the reference marsh flora in about 14 years. However, there were large variations in similarity during this 11-year period, which are directly explained by the pattern of rapid turnover early in the development period, followed by a period of slower change. Hence, the rate of convergence in composition of the restored and reference floras will probably be protracted beyond 14 years.

Use of Reference Sites

Besides providing information on the successional "end point," the reference site was useful in documenting annual variability. We were somewhat surprised at the degree of dissimilarity between one year and the next in the vegetation at what appeared to be a very stable and mature system. The unweighted and weighted similarity index values between successive years averaged 78 and 71% and ranged 70 to 89% and 59 to 84%, respectively. This has implications for mitigation performance monitoring programs (Kentula et al. 1992; Thom & Wellman 1996). These programs need to judge development based on data taken during the same year at both the restored site and the reference sites. Comparing the restored system in one year with a reference site from another year may affect conclusions. In addition, because the reference and restored sites exhibited considerable between-year variation, a restored site may only be expected to reach some "close approximation of the natural site," which coincides with the definition of aquatic ecosystem restoration provided by the National Research Council (1992). We suggest that the long-term average between-year similarity at the reference site (78% for the unweighted similarity in this study) might reasonably represent a "close approximation" similarity level to use as a goal for similarity between a restored site and its reference.

Monitoring Parameters

Overall the use of species composition and species cover provided a wealth of information on system development. These parameters typically show quickest recovery rates as compared with belowground param-

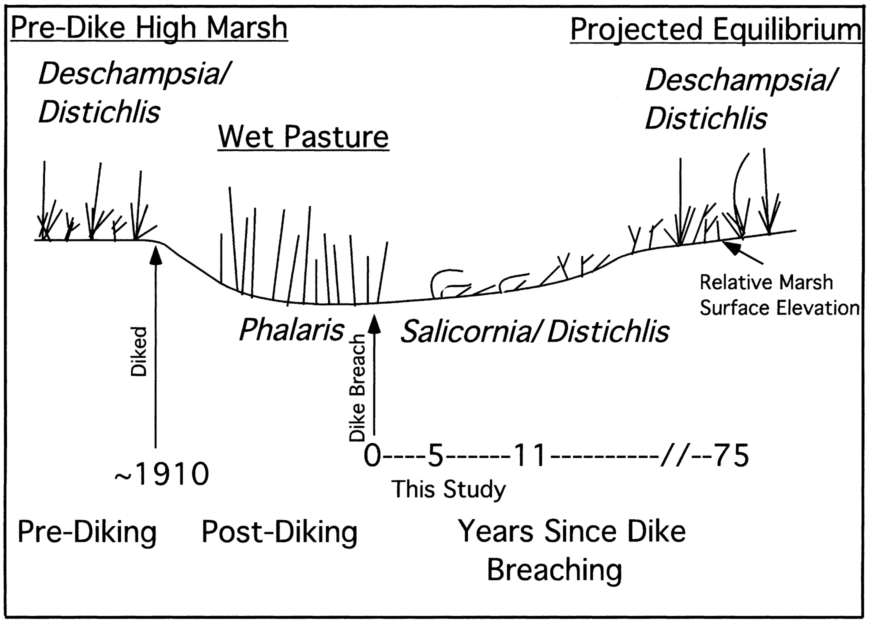


Figure 10. Conceptual model of recovery of the Elk River marsh system. Time line is based on accretion rate data from the restored marsh system (Thom 1992).

eters such as soil organic matter (Morgan & Short 2002, this issue; Roman et al. 2002, this issue; Warren et al. 2002, this issue).

We noted other changes in the system that can provide additional information. For example, channel formation has been proceeding since the dike was opened. This is important in that the channels allow fish access to the marsh at high tides (Gray et al. 2002, this issue). Common wildlife indicators (e.g., scat) have shown that the system is receiving use by a variety of species normally using tidal marshes (e.g., Great Blue Heron, river otter; R. Zeigler, personal observation).

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LITERATURE CITED

Armstrong, J. W., R. M. Thom, and K. K. Chew. 1980. Impact of a combined sewer overflow on the abundance, distribution and community structure of subtidal benthos. Marine Environmental Research **4:**3–23.

Atwater, B. F. 1987. Evidence for great Holocene earthquakes along the outer coast of Washington state. Science **236:**942–944.

Boumans, R. M. J., D. M. Burdick, and M. Dionne. 2002. Modeling habitat change in salt marshes after tidal restoration. Restoration Ecology **10:**543–555.

Bray, J., and J. Curtis. 1957. An ordination of the upland forest community of southern Wisconsin. Ecological Monographs **27:**325–349.

Callaway, J. C. 2001. Hydrology and substrate. Pages 89–117 in Joy B. Zedler, editor. Handbook for restoring tidal wetlands. CRC Press, Boca Raton, Florida.

Cooke, S. S. 1997. A field guide to the common wetland plants of western Washington & northwestern Oregon. Seattle Audubon Society, Seattle.

Cornu, C. E., and S. Sadro. 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's south slough. Restoration Ecology **10:**474–486.

Crooks, S., J. Schutten, G. D. Sheeron, K. Pye, and A. J. Davy. 2002. Drainage and elevation as factors in the restoration of salt marsh in Britain. Restoration Ecology **10:**591–602.

Eertman, R. H. M., B. A. Korman, E. Stikvoort, and H. Verbeek. 2002. Restoration of the Sieperda tidal marsh in the Scheldt estuary, the Netherlands. Restoration Ecology **10:**438–449.

Eilers, H. P. 1975. Plants, plant communities, net production and tide levels: the ecological biogeography of the Nehalem salt marshes, Tillamook County, Oregon. Ph.D. Dissertation. Oregon State University, Corvallis.

Frenkel, R. E., and J. C. Morlan. 1990. Restoration of the Salmon River salt marshes: retrospective and prospect. Final Report to U. S. Environmental Protection Agency, Region 10, Seattle. Oregon State University, Corvallis.

Gray, A., C. A. Simenstad, D. L. Bottom, and T. J. Cornwell. 2002. Contrasting functional performance of juvenile salmon habitat in recovering wetlands of the Salmon River Estuary, Oregon, U.S.A. Restoration Ecology **10:**514–526.

Kentula, M. E., R. P. Brooks, S. E. Gwin, C. C. Holland, A. D. Sherman, and J. C. Sifneos. 1992. An approach to improving decision making in wetland restoration and creation. EPA/600/R-92/150. U. S. Environmental Protection Agency, Washington, DC.

Morgan, P. A., and F. T. Short. 2002. Using functional trajectories to track constructed salt marsh development in the Great Bay Estuary, Maine/New Hampshire, U.S.A. Restoration Ecology **10:**461–473.

National Research Council. 1992. Restoration of aquatic ecosystems: science, technology, and public policy. National Academy Press, Washington, DC.

Nesbit, D. M. 1885. Tide marshes of the United States. Miscellaneous special report No. 7. U. S. Department of Agriculture, Washington, DC.

Prentice, S. A., and J. M. Kain. 1976. Numerical analysis of subtidal communities on rocky shores. Estuarine and Coastal Marine Sciences **4:**65–70.

Roman, C. T., K. B. Raposa, S. C. Adamowicz, M.-J. James-Pirri, and J. G. Catena. 2002. Quantifying vegetation and nekton response to tidal restoration of a New England salt marsh. Restoration Ecology **10:**450–460.

Simenstad, C. A., and R. M. Thom. 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecological Applications **6:**38–56.

Tanner, C. D., J. R. Cordell, J. Rubey, and L. Tear. 2002. Restoration of freshwater intertidal habitat functions at Spencer Island, Everett, Washington. Restoration Ecology **10:**564–576.

Thom, R. M. 1987. The biological importance of Pacific Northwest estuaries. Northwest Environmental Journal **3:**21–42.

Thom, R. M. 1992. Accretion rates of low intertidal salt marshes in the Pacific Northwest. Wetlands **12:**147–156.

Thom, R. M., and L. Hallum. 1991. Historical changes in the distribution of tidal marshes, eelgrass meadows and kelp forests in Puget Sound. Pages 302–313 in Proceedings of Puget Sound Research ’91. Puget Sound Water Quality Authority, Olympia, Washington.

Thom, R. M., and K. F. Wellman. 1996. Planning aquatic ecosystem restoration monitoring programs. IWR Report 96-R-23. Institute for Water Resources, U.S. Army Corps of Engineers, Alexandria, Virginia.

Thom, R. M., and T. B. Widdowson. 1978. A resurvey of E. Yale Dawson’s 42 intertidal transects on the Southern California mainland after 15 years. Bulletin Southern California. Academy of Sciences **77:**1–13.

Thom, R. M., A. B. Borde, and D. L. Woodruff. 2000. Tidal wetland plants distribution and primary control factors in Commencement Bay. Final report to NOAA Damage Assessment and Restoration Center Northwest, Seattle, Washington.

Warren, R. S., P. E. Fell, R. Rozsa, A. H. Brawley, A. C. Orsted, E. T. Olson, V. Swamy, and W. A. Niering. 2002. Salt marsh restoration in Connecticut: 20 years of science and management. Restoration Ecology **10:**497–513.

Weinmann, F., and K. Kunz. 1994. Wetlands: infusion of restoration into the management formula. Pages 1–17 in M. Martz, A. Jarvela, K. Kunz, C. Simenstad, and F. Weinmann, editors. Partnerships & opportunities in wetland restoration, Proceedings of a workshop, Seattle, Washington. U.S. Environmental Protection Agency, Seattle.

Williams, P. B., and M. K. Orr. 2002. Physical evolution of restored breached levee salt marshes in the San Francisco Bay estuary. Restoration Ecology **10:**527–542.

Zedler, J. B. 2001. Introduction. Pages 1–37 in J. B. Zedler, editor. Handbook for restoring tidal wetlands. CRC Press, Boca Raton, Florida.