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Report

Marsh Creation in a Northern Pacific Estuary: Is Thirteen Years of Monitoring Vegetation Dynamics Enough?

Neil K. Dawe[1], *Gary E. Bradfield*[2], *W. Sean Boyd*[1], *Donald E. C. Trethewey*[1], *and A. Nana Zolbrod*[2]

[1]***Canadian Wildlife Service***; [2]***University of British Columbia***

ABSTRACT

Vegetation changes were monitored over a 13-yr period (1982-1994) in the Campbell River estuary following the development of marshes on four intertidal islands. The marshes were created to mitigate the loss of a natural estuarine marsh resulting from the construction of a dry land log-sorting facility. Plant species coverage was measured along 23 permanent transects in planted and unplanted blocks on the constructed islands, and in naturally occurring low-marsh and mid-to-high marsh reference communities on nearby Nunn′s Island. Five dominant species, *Carex lyngbyei, Juncus balticus, Potentilla pacifica, Deschampsia caespitosa,* and *Eleocharis palustris* established successfully and increased in cover in both planted and unplanted areas. The planted, unplanted, and Nunn′s Island low-marsh sites had similar total plant cover and species richness by the 13th year. Principal components analysis of the transects through time indicated successful establishment of mid-to-low marsh communities on the constructed islands by the fourth year. Vegetation fluctuations on the constructed islands were greater than in the mid-to-high and low-marsh reference communities on Nunn′s Island. Results showed that substrate elevation and island configuration were major influences on the successful establishment and subsequent dynamics of created marsh communities. Aboveground biomass estimates of marshes on the created islands attained those of the reference marshes on Nunn′s Island between years 6 and 13. However, *Carex lyngbyei* biomass on the created islands had not reached that of the reference marshes by year 13. Despite the establishment of what appeared to be a productive marsh, with species composition and cover similar to those of the reference marshes on Nunn′s Island, vegetation on the created islands was still undergoing changes that, in some cases, were cause for concern. On three of the islands, large areas devoid of vegetation formed between years 6 and 13, probably a result of water ponding. Adaptive management has allowed us to modify the island configuration through the creation of channels to drain these sites in an attempt to reverse the vegetation dieback. These changes, occurring even after 13 years, further underscore the need for caution when considering the trading of existing natural, healthy, productive wetlands for the promise of created marshes that may or may not prove to be equal to the natural systems. Where marsh creation is warranted, we recommend that management of created marshes be adaptive and flexible, including a long-term monitoring program that should continue at least until the annual variation in vegetation of the created marsh is similar to that of natural, nearby systems.

KEY WORDS: adaptive management of ecosystems, *Carex lyngbyei*, *Deschampsia caespitosa*, *Eleocharis palustris*, estuarine marsh creation, *Juncus balticus*, long-term vegetation dynamics, mitigation cautions, natural vs. constructed wetlands, *Potentilla pacifica*, restoration ecology, wetland creation.

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INTRODUCTION

Restoration ecology involves the development of structural or functional characteristics of ecosystems that have been lost (Cairns 1988). It includes habitat creation that aims to establish plant communities that are representative of the original, undamaged state (Buckley 1989). Many restoration projects are implemented as mitigation for the loss of natural marshes resulting from development. In some cases, resource agencies have adopted a "no net loss" policy that requires the proponent to create replacement marsh habitat in some ratio of created to lost habitat, such as 2:1. Although this may seem reasonable, it assumes that enough is known about natural wetland systems that they *can* be created with all of their attendant functions. Because of the belief that we can build wetlands based on past, apparently successful attempts, it has also meant that the resource agencies often approve development proposals that destroy diverse, productive wetlands, rather than simply protecting the integrity of these natural wetland systems. Unfortunately, the apparently successful attempts have often been monitored for only a short time period (usually 1-2 years), if at all.

Coastal marsh restoration on the east coast of the United States has occurred to a far greater extent than on the Pacific coast, with mixed success (Seneca et al. 1976, Broom et al. 1986, Mason and Slocum 1987). On the U.S. Pacific coast, marsh restoration and creation projects have been carried out from San Diego to the Pacific Northwest (e.g., Zedler 1984, Frenkel and Morlan 1991, Zedler and Langis 1991, Simensted and Thom 1996). In British Columbia, many coastal marshes are in need of restoration because they have suffered heavy damage from past human activities, and are critical habitats for sustaining healthy populations of fish, migratory birds, and other wildlife (Butler et al. 1989, Levings and Macdonald 1991). Recently, many restoration projects have been undertaken in the estuaries of southwestern British Columbia. They have involved both restoration of degraded marshes and creation of new marsh habitat, principally as mitigation for development (Anonymous 1995).

On Vancouver Island, studies of vegetation change following the breaching of sea dikes to restore natural tidal flushing have been reported by Dawe and Jones (1986), Campbell and Bradfield (1988), and Dawe and McIntosh (1993). On the adjacent mainland, in the Fraser River estuary, several projects aimed at increasing the area of intertidal marshland have been undertaken since the early 1980's (Kistritz 1995, Adams and Williams, *in press*). Varying degrees of success were achieved, but even in the "successful" cases, species diversity and total vegetation cover remain below those of comparable, naturally occurring marshes. Improper design features relating to surface elevation and exposure to strong currents, as well as intense grazing by Canada Geese, were the major reasons that the created marshes failed.

Because our knowledge of these complex systems is poor (cf., Holling and Meffe 1996, Lee 1999), the adoption of a conservative management approach is warranted. Holling and Meffe (1996) suggest that the default condition of ecosystem management, unless proven otherwise, should be the retention of the natural state rather than any manipulation of system components or dynamics. When manipulation is deemed necessary, adaptive management should be encouraged. Adaptive management is a structured, continual process of "learning by doing," (Holling 1978, Walters 1986, Christensen et al. 1996). Management objectives are clearly established based on the best available science, and the results are monitored to provide timely feedback to managers, who may then modify their activities accordingly. As Hilborn (1992) notes, "If decisions are not made based on the results of monitoring and evaluation, learning will not take place."

Monitoring over the long term is an important component of adaptive management. Time scales for system components to respond to management prescriptions are typically long (Lee 1999) because they often deal with slowly changing variables. Also, systems are continuously changing through natural processes, making it difficult to determine their "normal state." As a result, our knowledge of them is always incomplete (Walters and Holling 1990, Ehrenfeld 1991). In addition to the natural variation in these systems, there are the added changes they undergo as the effects of human activities, which include the management activities, interact with them. It then becomes difficult to distinguish the effects of management from those of natural variation in the system (Walters and Holling 1990). Thus, ecosystems are "moving targets," having uncertain and unpredictable futures (Holling 1978, Walters 1986).

One of the largest marsh creation projects undertaken in British Columbia occurred in the Campbell River estuary in 1981 (Brownlee et al. 1984). The project involved the construction of four intertidal islands with a total plantaed surface area of 2.4 ha. Over 23,000 marsh vegetation plugs salvaged from the construction of a nearby log-sorting facility and holding pond were transplanted to the islands. An important adaptive component of the project was a long-term monitoring study of the vegetation growth on the islands. Although small-scale projects such as this have not received as much attention in the literature as have large-scale issues (e.g., management of a fishery), Johnson (1999) notes that there are many applications for an adaptive management approach, including wetland restoration.

Our paper summarizes the results of a monitoring study of the developing marsh communities in the Campbell River estuary conducted over a 13-yr period. The objectives were: (1) to evaluate the success of the marsh creation project when compared with a nearby, natural system and to be able to react to unforseen problems that may arise; and (2) to determine the degree to which the vegetation changes through time were related to substrate elevation, planting design, and island configuration. Long-term comparative studies such as this have rarely been reported for intertidal marshes along the northern Pacific coast (Race 1985); the results will be useful for other rehabilitation efforts in these ecosystems.

METHODS

Study area

The Campbell River estuary is located on the central eastern coast of Vancouver Island (50° 02′ N, 125° 15′ W; Figs. 1 and 2). Much of the estuarine habitat is situated behind Tyee Spit, which supports an area of light industry (Fig. 1). The estuary also has two marinas plus an active float plane base. From a watershed covering 1461 km^2, the river has the third largest discharge on the east coast of Vancouver Island (mean: 108 m^3 /s). The river flows have been regulated since 1949 by the John Hart dam.

Fig. 1. Aerial photo of the Campbell River estuary looking south. Shown are the constructed islands in relation to Tyee Spit, Nunn′s Island, and the dry land log-sort area (6 August 1998).

The mean and greatest tidal ranges in the estuary are 2.9 m and 4.6 m, respectively; there are two complete tidal oscillations daily (Bell and Thompson 1977). Temperature and precipitation data for the Campbell River area show that the first part of the growing season (April - June) tends to be cooler and wetter (mean monthly temperature 14.7ºC; mean monthly precipitation 51.4 mm) than the latter part of the growing season (15.5ºC; 39.8 mm; July-September). Dormant season months (October-March) tend to be cool and wet (6.2ºC, 160.3 mm; Anonymous 1993).

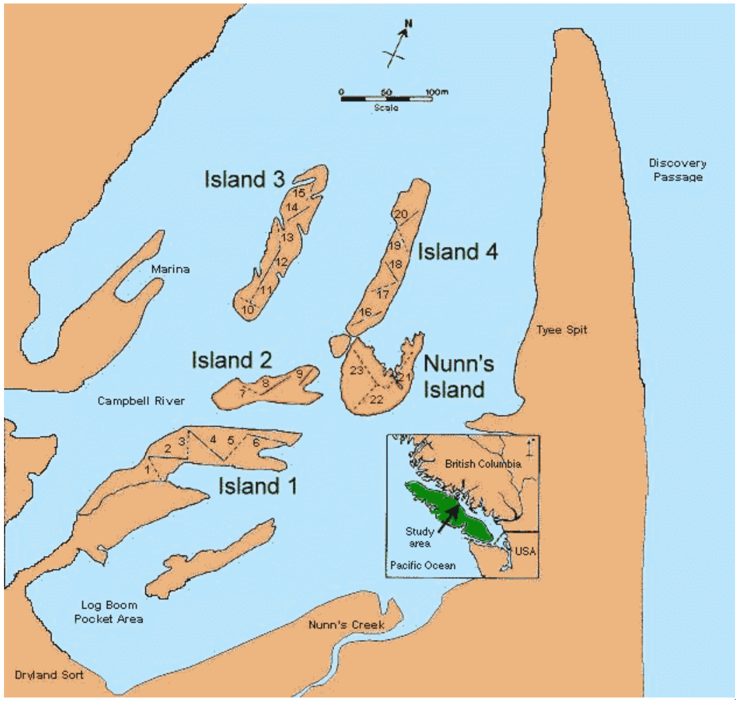
Between 1904 and 1981, the estuary was used primarily as a log storage and sorting area. As a result, the benthos became completely covered with woody debris and waste and, thus, was no longer biologically productive habitat. In 1981, the intertidal area used for storage and sorting of logs was reduced from 32.8 ha to 6.8 ha (Brownlee et al. 1984) because of the construction of a dry land log-sorting facility.

The Campbell River estuary provides important habitat for five species of Pacific salmon (*Onchorhynchus* spp.), steelhead (*O. mykiss*) and cutthroat (*O. clarki*) trout, and Dolly Varden char (*Salvelinus malma*; Brownlee et al. 1984), and supports both migrant and wintering populations of migratory birds (Dawe et al. 1995).

Experimental design and planting treatments

The marsh construction project was undertaken in 1981 as part of an agreement requiring British Columbia Forest Products Ltd. (now TimberWest Forest Ltd.) to compensate for the loss of about 0.42 ha of intertidal marshes through the expansion of their log-sorting facilities. Scientific aspects of the project, including island design and transplanting methods, were overseen by biologists from Fisheries and Oceans Canada and the Canadian Wildlife Service of Environment Canada. Four islands were constructed in the winter of 1981-1982 from dredge spoils recovered from the creation of the dry land log-sort holding pond. The islands produced a total surface area (excluding side slopes) of about 2.4 ha (Fig. 2). The islands were designed to optimize planting success, taking into consideration elevation, slope, and substrate stability (Brownlee et al. 1984). Substrate elevations of islands 1, 2, and 4 were graded to approximately 3.0 m above chart datum (ACD), considered to be optimum for growth of *Carex lyngbyei* in the estuary. *C. lyngbyei* is a productive sedge with high habitat values for juvenile fish and migratory birds. Island 3 was graded from subtidal to about 4.0 m ACD, which was similar to the elevational gradient on Nunn′s Island, the naturally occurring reference island in the estuary.

Fig. 2. Map of the study area showing the four constructed islands and Nunn's Island. Lines and numbers on the islands indicate transects; on the constructed islands, solid lines denote transects through transplanted areas and broken lines denote transect through unplanted areas. The inset shows the regional location of the study area in coastal British Columbia.



Over 23,000 marsh "plugs" containing substrate, plant rhizomes and roots, and aerial shoots were placed into planting blocks (size range: 300 m^2 to 1,100 m^2) on the constructed islands during the nocturnal, low tides of February 1982. The plugs measured approximately 15 x 15 x 20 cm deep. They were cut from larger mats collected during the previous November from a donor site in the southeastern part of the estuary, and were moved to an overwintering site near island 1. The plugs were planted at two randomly selected densities in the planting blocks on each island: 0.5-m or 1.0-m intervals (Table 1). Some areas on the islands were left unplanted to assess the extent and rate of natural colonization. In total, about 0.8 ha were planted on the four constructed islands (Brownlee et al. 1984).

Table 1. Mean elevation (ACD is above chart datum), treatments, and inundating water salinities of permanent transects on marsh transplant islands in the Campbell River estuary, British Columbia.

Transect	Elevation (m ACD); mean (1 SD)	Treatment	Water column surface salinity (‰)	Water column bottom salinity (‰)
Island 1				
1	3.06 (0.06)	unplanted		
2	3.14 (0.09)	planted, 1 m^2	5	5
3	3.36 (0.01)	unplanted		
4	3.25 (0.08)	planted, 1 m^2	4	6
5	3.15 (0.06)	unplanted		
6	3.08 (0.09)	planted, 0.5 m^2	3	13
Island 2				

7	2.89 (0.08)	unplanted		
8	3.08 (0.09)	planted, 1 m^2	3	3
9	3.09 (0.02)	planted, 0.5 m^2	4	9
Island 3				
10	3.96 (0.58)	unplanted		
11	4.19 (0.08)	unplanted		
12	3.05 (0.15)	planted, 0.5 m^2	5	13
13	2.59 (0.32)	unplanted		
14	2.45 (0.23)	planted, 0.5 m^2	5	25
15	2.48 (0.14)	planted, 0.5 m^2	5	24
Island 4				
16	3.17 (0.12)	planted, 0.5 m^2	4	7
17	2.85 (0.09)	unplanted		
18	2.94 (0.16)	planted, 0.5 m^2	5	5
19	2.96 (0.13)	unplanted		
20	2.74 (0.13)	planted, 1 m^2	5	14
Nunn′s Island				
21	2.91 (0.14)	low marsh	4	6
22	3.58 (0.23)	mid-to-high marsh	5	5
23	3.58 (0.20)	mid-to-high marsh		

Unfortunately, neither species cover nor elevations at the donor site (which was destroyed in creating the log-holding pond) were recorded before the vegetation mats were cut and moved to the overwintering site. However, the dominant species noted at the donor marsh included *Carex lyngbyei*, *Juncus balticus, Equisetum* spp., *Potentilla pacifica,* and *Deschampsia caespitosa* (Brownlee et al. 1984). With the exception of *Equisetum* spp., these species are typical of low-salinity, estuarine marshes in south-coastal British Columbia. The species composition of the planted material suggests that the donor site elevation was in the mid-marsh range (Dawe and White 1982).

Sampling

In late June 1982, 23 permanent transects (30-80 m in length) were established, running diagonally through the transplanted and unplanted blocks (Fig. 2 and Table 1). The transects were surveyed to facilitate their location in future years, and the end points of each transect were marked with metal rods driven into the substrate. Two types of reference marshes on Nunn′s Island—a "low" marsh (transect 21), and a "mid-to-high" marsh (transects 22 and 23)—were used to evaluate the success of the constructed marshes.

Sampling surveys of all transects were carried out during late June or early July of 1982, 1983, 1984, 1985, 1986, 1988, and 1994. For the first two years, the success of individual transplant plugs was monitored; by 1984, it became too difficult to distinguish the individual plugs over much of the study area. Plugs were considered successful if they contained at least one living vascular plant.

Vegetation data were collected by systematically sampling 1.0 m^2 relevés spaced at 5.0-m intervals along the permanent transects as described in Dawe and White (1982, 1986). Cover classes of vascular plant species were recorded using the Braun-Blanquet cover-abundance scale (Mueller-Dombois and Ellenberg 1974). Although the Braun-Blanquet scale tends to minimize differences between observers in cover estimates, such differences may arise, particularly when the actual cover is close to the class boundaries (i.e., 25%, 50%, 75%). In this study, the senior author participated in all sampling surveys to ensure consistency. Nomenclature follows Hitchcock and Cronquist (1973).

Transects 10 and 11 on island 3 were not sampled in 1994, as they were considerably higher in elevation than the other transects, and were developing into upland communities, not representative of intertidal marsh vegetation.

In 1988 and 1994, estimates of aboveground biomass were obtained. Five 0.125-m^2 vegetation samples were randomly taken from each of the planted and unplanted blocks and the Nunn′s Island reference marshes. All living vegetation was clipped with garden shears at substrate level within the sample frame, and was placed in opaque plastic bags. The collected samples were stored in a freezer until they could be analyzed.

In the laboratory, the individual samples were processed as outlined in Eilers (1975). Briefly, they were separated into their constituent species fractions, the number of stems of each species was determined, and the species fraction was weighed to the nearest 0.1 g on a triple beam balance. The species fraction was then cut into small pieces and placed in an open paper bag in a ventilated drying oven set at 85°C. Subsamples were taken where the species fraction exceeded about 50 g. The samples and subsamples were dried until constant mass was attained; subsample mass values were converted to total dry mass by using a simple proportion calculation.

On the high tide of 26 June 1984, the salinity of the inundating waters over each block was determined using a conductivity meter. Measurements were gathered to determine if salinity of the inundating waters could be a factor in the poor growth of the vegetation at the lower elevations of island 3.

Data analysis

Vegetation dynamics were analyzed using principal components analysis (PCA), which allowed us to summarize and graph the changes in vegetation over time. Problems have been associated with PCA of highly variable data sets, but the method is effective where compositional gradients are less extreme (Gauch 1982), as is the case with intertidal marsh vegetation. The PCA was based on a species covariance matrix calculated from the mean cover class data for transects from all sampling periods. Mean cover class was determined by summing the cover class values of species in each transect, and then dividing by the number of relevés examined. Mean cover across the transect is more suited for analyzing variation at the community scale, where the absence as well as occurrence of species is of interest. This allowed a comparison of the vegetation changes over time among transects on the separate islands. Interpretations of the first two PCA axes were derived from correlations of the factor scores with the transect means of species cover, species richness, total cover, and elevation. The elevation correlations were determined using the 1988 PCA transect scores, as this was the last year in which transects 10 and 11 were surveyed. The software program SYSTAT was used for all calculations.

Changes in species composition and cover over time were also calculated separately for the various islands and treatments (i.e., planted blocks, unplanted blocks, and Nunn′s low- and mid-to-high reference marshes). Best estimates of mean cover-abundance for each species were calculated by summing the midpoints of each species′ Braun-Blanquet scale range (i.e., by setting *r* = 0.01,+= 0.5, 1 = 3, 2 = 15, 3 = 37.5, 4 = 67.5, 5 = 87.5) and dividing by the number of occurrences of the species within the data set. An estimate of total mean cover was then calculated by summing the mean cover values of each species within the data set. These data were used to compare mean cover between treatments and islands. The method tends to overestimate cover somewhat, but the results are relatively comparable. The frequency of occurrence of each species was calculated by dividing the number of occurrences of a species within the data set by the total number of relevés within the data set. Mean species richness values by treatment and by island were calculated by summing the total number of species in each transect and dividing by the number of transects in each treatment or island. We applied the cell-means ANOVA approach to the aboveground biomass data to test for differences between treatments and between islands.

RESULTS

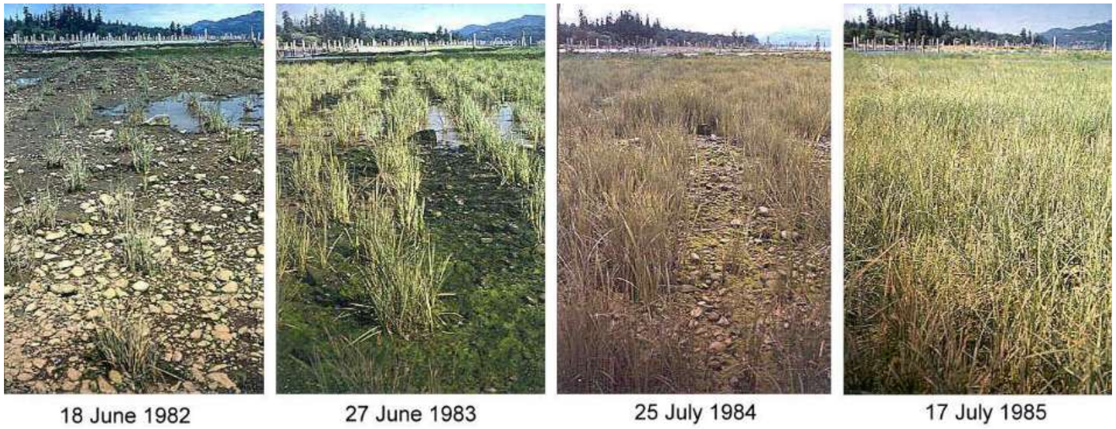
Island substrate stability

During the autumn and winter of 1982, the islands in the estuary were subjected to high river discharges at low tides; however, there was no apparent damage or destabilization to the created islands (Brownlee et al. 1984). Continued observations to 1994 suggested that the islands remained stable, with no serious erosion occurring, even on island 3 directly adjacent to the main river channel.

Vegetation plug success

Plug success on the four constructed islands was 92% by July 1982. Most plug mortality occurred in the two low-elevation blocks of island 3 (transects 14 and 15). Plug mortality on island 3 increased from 13% in July 1982 to 28% in August 1983, and vegetation growth of the two low-elevation blocks never recovered. By August 1983, overall plug success on the four islands was about 90%. By 1984, it became too difficult to differentiate individual plugs among the coalescing vegetation on many of the blocks (Fig. 3).

Fig. 3. Planted block C, row 10 on island 1 (transect 4), Campbell River estuary, 1982 to 1985. By 1984, the vegetation had spread so that recognition of individual plugs was impossible in some areas. By 1985, individual plugs were unrecognizable throughout the most of the study area.



By June 1982, two plant species had exhibited rapid spreading characteristics: *Eleocharis palustris,* by rhizomes, and *Potentilla pacifica,* by stolons (Fig. 4). Two other species, *Scirpus cernuus* and *Scirpus validus,* were also colonizing some of the islands by July 1982. By the second growing season (July 1983), *Lilaeopsis occidentalis* also began to dramatically expand away from the plugs and cover the bare substrate.

Fig. 4. *Eleocharis palustris,* spreading by rhizomes, and *Potentilla pacifica,* spreading by stolons, exhibited favorable colonization and substrate stabilization qualities on the Campbell River estuary (27 June 1983).

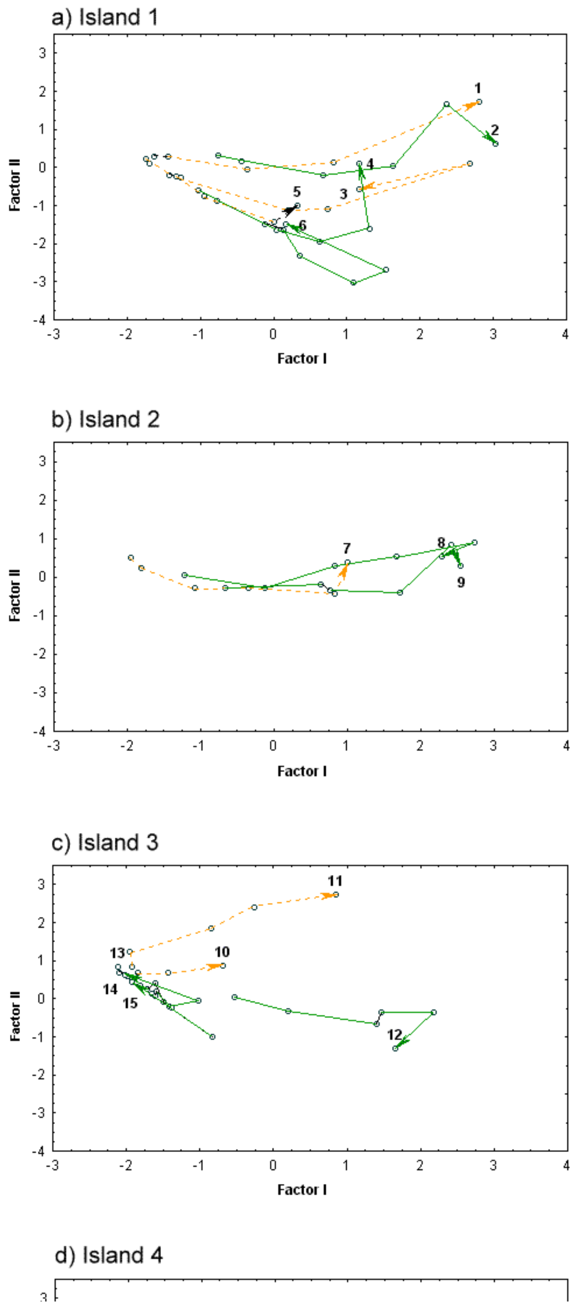
Species composition

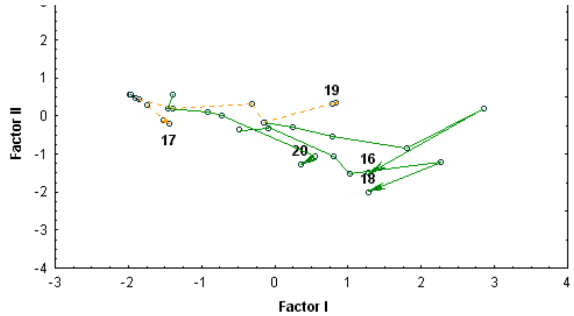
We identified 29 vascular plant species in the plugs 3 mo after planting, with four species dominating in overall frequency: *Carex lyngbyei* (in 76% of all plugs), *Eleocharis palustris* (55% of all plugs), *Potentilla pacifica* (42%), and *Juncus balticus* (39%). Subsequently, 60 vascular plant species were documented at the Campbell River study site over the 13-yr period (Appendix 1). Six species, all widespread in intertidal marshes along the northeastern Pacific coast at this latitude, were common at all treatment and reference marsh sites: *Carex lyngbyei, Juncus balticus, Potentilla pacifica, Deschampsia caespitosa, Eleocharis palustris,* and *Lilaeopsis occidentalis.* Species found only in the Nunn′s mid-to-high marsh included *Sium suave, Vicia americana, Daucus carota, Dodecatheon pulchellum, Mentha arvensis, Platanthera greenei, Sisyrynchium angustifolium, Hypericum formosum, Fritillaria camschatcensis,* and *Ranunculus occidentalis.* This increased diversity relates to the less frequent flooding, lower salinity of the inundating waters, and greater organic content of soil at the higher elevations on Nunn′s Island (Dawe and White 1982). Two species, *Plantago maritima* and *Typha latifolia,* were found only on the planted blocks for one or two years. *Glaux maritima* was found only on the unplanted blocks in one year. Twenty species, primarily upland species, were recorded only along transects 10 and 11 at the higher elevations of island 3.

Plant community development

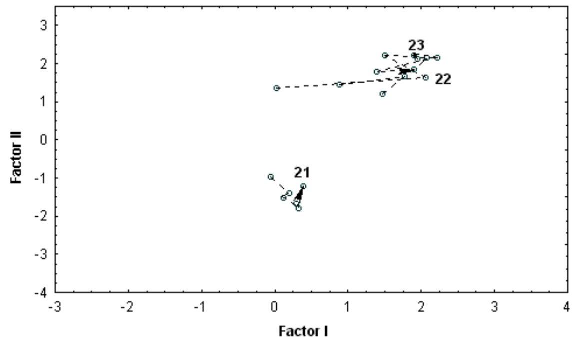
The vegetation dynamics at the scale of the developing plant communities is depicted by the changing positions of transects in the plane of the first two PCA axes (Fig. 5). The first PCA axis has a strong positive correlation with four of the dominant species (*Carex lyngbyei, Juncus balticus, Potentilla pacifica,* and *Deschampsia caespitosa*), and a strong negative correlation with bare ground (no cover). Thus, axis 1 represents a gradient of increasing vegetation cover. Axis 2 is positively correlated with elevation and, correspondingly, with increasing species richness such as that found in the mid-to-high marsh on Nunn′s Island. Species positively correlated with axis 2 include *Aster subspicatus, D. caespitosa,* and *Trifolium wormskjoldii.* Species negatively correlated with axis 2 include *Carex lyngbyei, Eleocharis palustrus,* and *Lilaeopsis occidentalis,* all of which have greater cover at the lower elevations. Thus, the second axis reflects the marsh platform gradient running from the low-elevation, *Carex*-dominated marsh, to the higher diversity, mid-to-high elevation marshes, as represented on Nunn′s Island.

Fig. 5. Transect trajectories on PCA axes 1 (horizontal) and 2 (vertical) based on the PCA factor scores. The two axes explain 41% and 22% of the variance, respectively, in species composition and cover. The trajectories denote the changes in vegetation occurring through time on the four constructed islands (islands 1-4; a-d) and the reference marsh on Nunn's Island (e). The islands have been graphed separately to emphasize interisland differences, although the PCA was performed on the combined data from all islands. Trajectories begin at the 1983 transect position, move through 1984, 1985, 1986, and 1988, and terminate at the 1994 position (except transects 10 and 11, which terminate in 1988), where the transect number and arrowhead are shown. For the created island plots (a-d), solid (green) lines denote transects through planted areas; broken (orange) lines denote transects through unplanted areas.





e) Nunn's Island



Most transects on the constructed islands showed an increase in plant cover over the study period, as indicated by their positive trajectories along PCA axis 1 (Fig. 5a-d). In contrast, the Nunn's Island transects remained relatively stable over the 13-yr study period (Fig. 5e). Several transects on islands 1, 2, and 4, showed decreased plant cover in 1994, with large areas devoid of vegetation (e. g., transects 3, 6, 8, 12, 16, 18, and 20). With the exception of transect 3, all were transects through planted blocks. In 1994, we noted for the first time some heavy grazing of the *Carex lyngbyei* plants by Canada Geese on portions of the low marsh (transect 21) on Nunn's Island and on transects 19 and 20 on island 4.

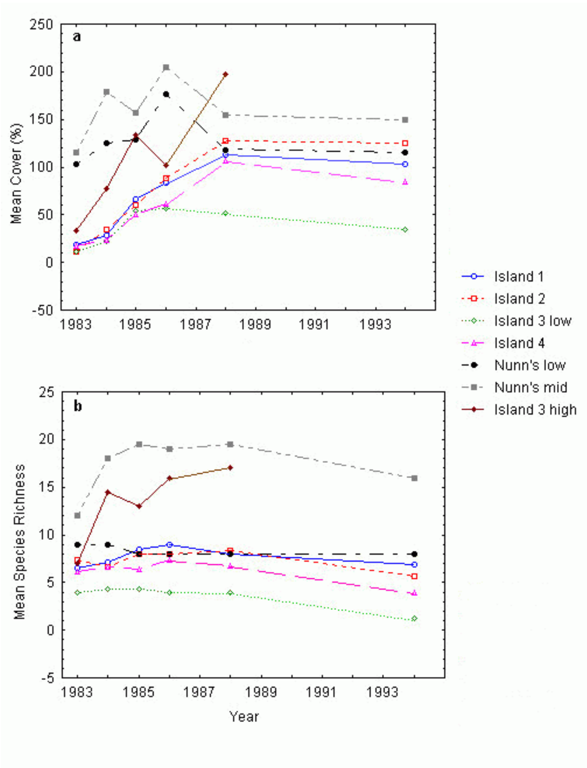
For most of the planted transects, species richness remained relatively constant or decreased through time, as indicated by the unchanging or negative trajectories along PCA axis 2. Moreover, interisland differences in vegetation dynamics appeared to be stronger than intertreatment differences. Only transect 4 on island 1 showed an increase in species richness between 1988 and 1994, and only from 7 to 8 species. The two high-elevation transects (10 and 11) on island 3 showed continuing increases in species richness to 1988.

The Nunn's Island transects (Fig. 5e) provide a reference for evaluating the vegetation changes on the four constructed islands. Nunn's low-marsh transect 21 was relatively stable over the 13-yr study period, remaining dominated by *Carex lyngbyei,* but also with some *Juncus balticus, Potentilla pacifica, Eleocharis palustris, Lilaeopsis occidentalis,* and *Deschampsia caespitosa.* Nunn's mid-to-high marsh transects 22 and 23 also were relatively stable in terms of cover and frequencies of the dominant species, and have consistently retained a species-rich, high-cover plant community.

Comparisons among islands

Mean percent cover was highest on the Nunn's mid-to-high marsh, followed by the high-elevation block of island 3, island 2, and Nunn's low marsh (Fig. 6a). With the exception of island 3, mean percent cover increased through 1988 and then showed similar cover or slight decreases by the 1994 sampling. Mean species richness on the created islands was similar to the Nunn's low-marsh values to 1988; however, all islands, with the exception of Nunn's mid-to-high marsh and the high elevations of island 3, ended the study period with mean species richness values lower than the Nunn's low marsh (Fig. 6b).

Fig. 6. Changes in (a) mean cover class and (b) species richness by island from 1983 to 1994 on the Campbell River estuary. The high-elevation block of Island 3 (island 3 high) has been considered separately from the other lower elevation blocks on island 3.



Island 1 established the second highest plant cover and the most species-rich vegetation of all of the constructed islands, except the higher elevation transects 10 and 11 on island 3 (Fig. 6). Dominated by *Carex lyngbyei, Juncus balticus,* and *Potentilla pacifica,* the species richness and cover on island 1 was generally similar to that of Nunn's low marsh, although the high proportion of *Juncus balticus* along some of the island 1 transects had "moved" them more toward the Nunn's mid-to-high marsh (Fig 5a). Transect 1 was positioned higher than the others along the second PCA axis because of its high cover of *Juncus balticus* and low cover of i>Carex lyngbyei. Transects 5 and 6 were dominated by *Carex lyngbyei* and contained few other species, thus aligning themselves in the ordination close to Nunn's low-marsh transect 21 (Fig. 5e). During the 1994 sample period, the mean cover of transects 3, 4, and 6 decreased when large openings devoid of vegetation appeared in some of the blocks (Fig. 7 and Fig. 8; note the reversal of their trajectories in Fig. 5).

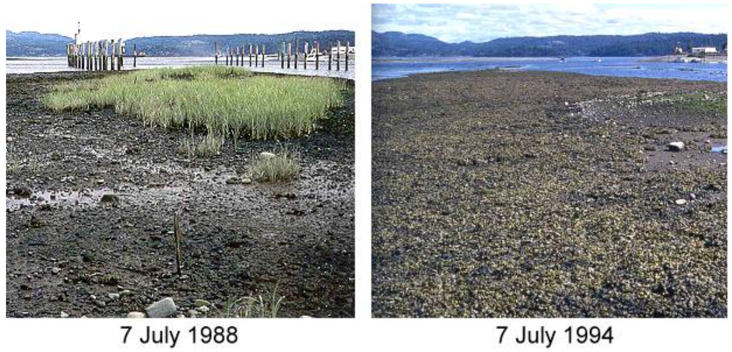
Fig. 7. Dieback in the vegetation of island 1, block C (transects 4 and 5), 7 July 1994 (left), and island 2, block A (transect 8), 7 July 1994. Vegetation covered most of these open areas in 1988; the vegetation dieback took 7-13 years to appear. This problem would probably have been prevented through better island design.

Fig. 8. Aerial photo of Nunn′s Island and three of the created islands (1, 2, and 4) on the Campbell River estuary, 6 August 1998. Arrows point to areas on the islands where vegetation dieback had occurred.

Transects on island 2, with the highest mean cover and second highest mean species richness of the constructed islands (Fig. 6), occupy a position in the ordination midway between the low- and mid-to-high marsh reference communities on Nunn′s Island (Fig. 5b). *Juncus balticus* became the dominant species along all three transects on island 2. Planted and unplanted sites increased in plant cover over the study period, with the exception of transect 8 (planted), which showed a decrease in 1994, resulting from large openings in the vegetation cover similar to those noted on island 1 (Figs. 5b, 7, and 8). The two planted transects had slightly higher plant cover, overall, than the unplanted transect.

Island 3 showed the greatest variability among transects of all of the constructed islands (Figs. 5c and 6). Only traces of vegetation appeared throughout the study on unplanted transect 13. On planted transects 14 and 15, the vegetation appeared to do well only at the slightly higher elevations of the transects, but by 1994, they had no vascular plant cover (Figs. 5c and Fig. 9). This accounted for the marked decrease in mean cover values for the lower elevations of this island (Fig 6a). Planted transect 12 had a mean elevation similar to that of the transects on island 4, and also showed similar vegetation dynamics, moving toward a *Carex lyngbyei*- and *Juncus balticus*-dominated mid-to-low marsh community with low species richness (Fig. 5c). Unplanted transects 10 and 11, on the other hand, increased in plant cover and species richness, but not with typical marsh species (Figs. 5c and 6). These transects were at a considerably higher elevation than the others, and were being invaded by species more characteristic of high marsh and upland sites.

Fig. 9. The low elevations of island 3 (transect 14 in foreground; transect 15 in background). By 1988, the surviving vegetation (*Carex lyngbyei*) only occurred on the higher elevation points of the transects By 1994, no vascular plant vegetation could be found. The higher inundating water salinity and longer inundation periods on these two low-elevation transects contributed to the failure of the vegetation growth. The pilings shown in the 1988 figure were cut and removed as part of a general clean-up of the estuary.



All transects on island 4 increased in plant cover through 1988, but subsequently showed decreases in cover and richness (Fig. 5d, Fig. 6, and Fig. 10). Openings in cover also occurred on this island (Fig. 8). The planted transects indicated development toward *Carex*-dominated, low-marsh vegetation, whereas transect 19 (unplanted), with higher *Juncus balticus* cover, did not follow this trend (Fig. 5d).

Fig. 10. Changes in vegetation growth over time on island 4, 1982 to 1994. Transect 16 is in the lower foreground of each photograph and transect 20 is in the upper background.

Comparisons among treatments

Year-to-year variation in mean cover was relatively small in the Nunn's low- and Nunn's mid-to-high transects, except for 1986; however, increases in total cover occurred in the planted and unplanted treatments, at least until 1988 (Fig. 11). Between 1988 and 1994, cover in both the planted and unplanted treatments did not differ, suggesting that some consistency had taken place between 1988 and 1994. By 1994, both planted and unplanted (Fig. 12) treatments had cover similar to that of the Nunn's low marsh.

Fig. 11. Changes in (a) mean cover class and (b) mean species richness by treatment from 1983 to 1994 on the Campbell River estuary. The high-elevation block of island 3 (island 3 high) has been considered separately from the other blocks on island 3. The data for Nunn′s Island are repeated from Fig. 6.

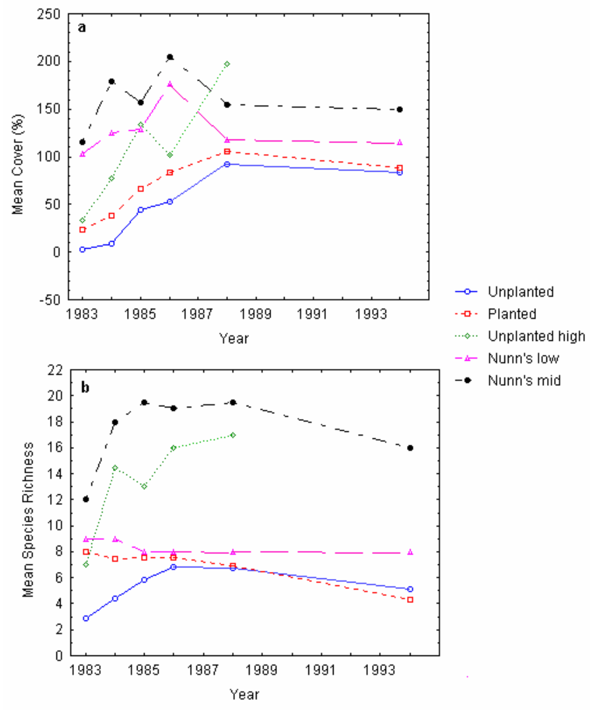


Fig. 12. Changes in vegetation over time on unplanted transect 1, island 1, Campbell River estuary, 1984 to 1994; planted transect 2 is located in the upper right background. By 1988, there was no difference between the aboveground biomass of the unplanted transects and the planted transects.

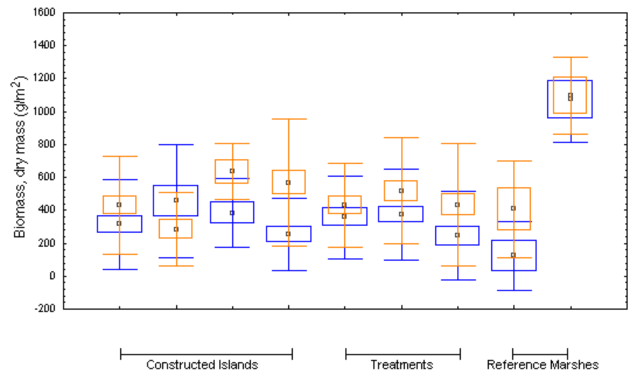
Species richness was highest in the Nunn's mid-to-high marsh, followed by the high-elevation, unplanted block of island 3, Nunn's low marsh, and then the planted blocks and unplanted blocks (Fig. 11). The planted blocks maintained a relatively constant species richness until 1986 and then began to drop, ending with lower mean richness than the unplanted transects. That was due primarily to the vegetation die-off of transects 14 and 15. The unplanted transects showed a continuous increase in species richness until 1986, after which their richness declined slightly.

Up to 1988, the planted and unplanted transects followed similar PCA trajectories, generally moving toward higher cover, similar to the Nunn's low-marsh community (Figs. 5 and 11). The exceptions were transects 10 and 11 on island 3, which increased in both cover and species richness, and unplanted transect 13, which supported only a trace of vegetation throughout the study. By 1988, transects 14 and 15, both traversing planted treatments on island 3, supported only *Carex lyngbyei;* by 1994, they were devoid of any vascular plant growth (Fig. 9). Another notable feature of the 1988-1994 period was the abrupt change in axis 1 trajectories of several of the planted transects, generally moving toward less cover, whereas the unplanted transects (except for transect 3) remained relatively stable or only slightly reduced their cover (Figs. 5a-d).

Biomass estimates

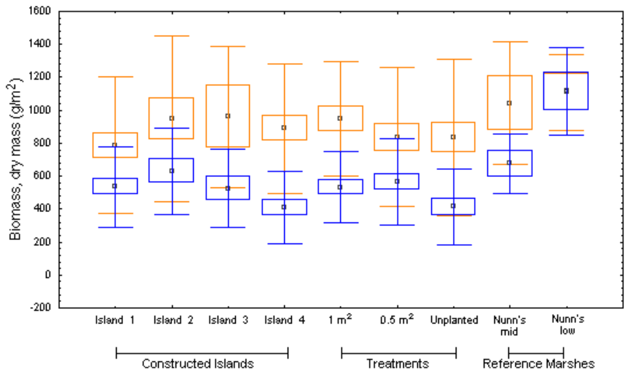
Total aboveground biomass estimates for *Carex lyngbyei* from the Nunn's Island low-marsh transect were greater ($P < 0.004$) than those of other treatments and other sites in both 1988 and 1994 (Fig. 13). Of interest, however, is the lack of significant difference in *Carex lyngbyei* biomass between the planted and unplanted blocks as early as 1988.

Fig. 13. *Carex lyngbyei* aboveground biomass (mean [dot] standard error [box] and standard deviation [line]) for islands and treatments on the Campbell River estuary for the years 1988 (blue) and 1994 (orange). Of interest is the lack of significant difference in *Carex lyngbyei*aboveground biomass between the planted and unplanted blocks as early as 1988.



Total aboveground biomass on the Nunn's Island low marsh was different from other sites and other treatments in 1988 ($P < 0.001$) (Fig. 14). However, by 1994, there was no difference in total aboveground biomass, either between sites or between treatments ($P > 0.30$).

Fig. 14. Total aboveground biomass (mean [dot] standard error [box] and standard deviation [line]) for islands and treatments on the Campbell River estuary for the years 1988 (blue) and 1994 (orange). Of interest is the lack of significant difference in total aboveground biomass between the planted and unplanted blocks as early as 1988.



By 1988, there was no difference between the 0.5-m and 1.0-m planted treatments and unplanted treatments for either the *Carex lyngbyei* biomass or the total biomass estimates ($P > 0.30$).

Inundating water salinities

Although the small sample size of salinities precluded statistical comparison (Table 1), it appeared that the lower elevation transects 14 and 15 on island 3 experienced higher salinities at the substrate level than did other transects at higher elevations (24-25 ‰ vs. 3-14 ‰), which, coupled with the longer inundation periods, could have been a contributing factor to the plug failures there.

DISCUSSION

The marsh creation project in the Campbell River estuary has resulted in the establishment of plant communities with typical intertidal marsh plant species at this latitude. Over the 13-yr study period, the constructed islands remained stable, with little apparent erosion. As a result, most of the planting blocks on the islands established *Carex lyngbyei*- or *Juncus balticus*-dominated communities with total cover similar to that of the natural low-marsh or mid-to-high marsh communities of adjacent Nunn′s Island. A striking result was the vegetation growth on the unplanted blocks that, between the 7th and 13th years, had reached species composition and aboveground biomass levels comparable to those of both the planted and the natural marshes. Thus, the project has achieved Zedler's (1988) marsh creation goals of establishing plant cover and species richness near natural levels.

Despite this establishment, however, the marsh communities on the constructed islands are not as stable as those on Nunn′s Island, as suggested by the movements of transects in the PCA ordination (Fig. 5). These changes are due more to year-to-year shifts in relative abundance and cover than to actual changes in species composition. Moy and Levin (1991) and Zedler (1988) note that, in general, coastal salt marshes are subject to high temporal variability and are inherently unstable. Our data from this system, however, suggest the opposite. The Nunn′s Island low- and mid-to-high marshes were, in contrast to the planted and unplanted blocks, relatively stable over the study period, as suggested by only minor relative movements within the PCA ordination.

Climatic variation undoubtedly affects year-to-year variation in marsh vegetation growth. Allison (1992) found that rainfall during both the dormant period and early growing season was correlated with vegetation changes in a California salt marsh. Sampling error, despite attempts to minimize the problem, could also contribute to the minor shifts in the ordination of the Nunn′s Island transects.

Noticeable marsh establishment on the constructed islands in the Campbell River estuary began to appear about four years after the initial planting in 1982. By 1988, most of the planted transects were similar in species composition to the Nunn′s low or mid-to-high marshes, but it was not until some time between 1988 and 1994 that species cover and frequency values attained values similar to those of the natural marshes. Vegetation in the unplanted blocks was initially slower to establish, but by 1988, cover and species richness values approached those of the planted blocks.

Broome et al. (1986) found that primary productivity levels of transplanted *Spartina alterniflora* marshes resembled natural marshes after four years, and stabilized thereafter. This was not the case in our study, however. Although the cover and species composition of the constructed marshes were similar to those of the Nunn′s low marsh six years after planting, total aboveground biomass estimates of the created marshes were only about 40% that of the Nunn′s low marsh; *Carex lyngbyei* biomass was even less, at about 25%. Thirteen years after planting, a significant difference in the total aboveground biomass estimates of the planted marshes vs. the Nunn′s Island marshes could not be detected. *Carex lyngbyei* biomass of the planted and unplanted sites, however, reached only about 40% that of the Nunn′s Island low-marsh *Carex* biomass.

Other marsh creation projects have had mixed results in establishing successful plant communities. In a marsh transplant experiment in the Fraser River estuary, Pomeroy et al. (1981) found that one of three sites had negligible plug survival, while at the other two sites, *Carex lyngbyei* plug survival was 65% and 84%, which is somewhat lower than our plug survival rate of 90%. In both of their successful transplant sites, however, transplant shoot heights were not as great as those in the donor sites. Moy and Levin (1991), in a study of

Spartina salt marsh creation in North Carolina, found that some constructed marshes achieved rates of primary productivity comparable to those of a natural marsh, but noted that salt marshes should not be considered a replaceable resource. Mason and Slocum (1987) found that nine of 19 constructed wetlands successfully established salt marsh vegetation in Virginia's coastal zone, while six more were apparently on the way to success.

Substrate elevation appears to be a major factor in the establishment of marsh communities on the Campbell River estuary. There, islands 1 and 2 had a higher elevational range than island 4 (Table 1) and, correspondingly, had a higher cover and richer plant communities. Island 4 vegetation was more closely aligned with Nunn′s low marsh in cover and richness, and was roughly at the same mean elevation as the Nunn′s low marsh. The least successful transects, with little or atypical plant establishment, were found on island 3. The low elevations on island 3 probably contributed to the poor plant establishment on transects 13-15, because of the relatively longer inundation periods and the higher salinities of the inundating waters. Transect 12, within the elevational range of the other island transects, was successful. Pomeroy et al. (1981) also found that elevation and the associated salinity effect were strong determinants of plug survival in Fraser marsh transplants. Dawe and White (1982) note that substrate elevation is probably the most agreed-upon factor in determining vegetation zonation in brackish and salt marshes.

The diversity of species in the plugs planted in 1982 appeared to have little effect on the species composition of the resulting plant communities. Most planted and unplanted areas became increasingly dominated by *Carex lyngbyei* or *Juncus balticus,* regardless of the species composition of the plug or the species proximate to the unplanted areas. The fact that the unplanted transects eventually reached higher mean species richness than the planted transects and showed less of a decline in cover between 1988 and 1994 may have had more to do with our moving the donor vegetation from its mid-marsh location to lower elevations on the islands, with their longer periods of inundation, than to any other factor. Seeds or propagules arriving at the unplanted areas, no matter what the elevation, would not be likely to establish unless they found site conditions suitable for colonization (van der Valk and Davis 1976).

These results suggest the importance of ensuring that donor vegetation is moved to a location where elevation and inundating water salinities are similar to those of the donor site. Although seeds or propagules may not establish if conditions are unsuitable, the vegetation in the plugs could appear to be doing well initially, even in less-than-suitable conditions. It has been our experience that marsh vegetation is quite resilient. Even when changes are thrust upon the plants that make conditions unsuitable, they often tend to ‟hang on″ for years. For example, *Carex* growth at the low elevations of island 3, much reduced in cover, but still extant six years after planting, had disappeared by 1994. Dawe and McIntosh (1993), found that *Carex lyngbyei* stands were relatively stable in terms of cover and frequency for the first five years of their marsh restoration project, but by year 10, the *Carex* stands had disappeared because of increased interstitial soil salinities. Simenstad and Thom (1996) also noted a precipitous decline in the shoot density of *Carex lyngbyei* on their study area seven years after transplantation; the decline occurred after a gradual, linear increase over the first six years. The importance of such slowly changing variables can only be appreciated if monitoring continues for the long term.

Other factors reported to influence the rate of plant establishment in marsh creation are substrate characteristics and sedimentation rates (Seneca et al. 1976), grading and erosion control (Mason and Slocum 1987), and proximity to a natural marsh for seeding (Mason and Slocum 1987, Zedler 1988). On the Fraser River estuary, the main factors leading to the failure of marsh transplant projects included unstable substrate, incorrect substrate elevations, saturated soils, driftwood accumulations, and grazing by Canada Geese (Adams and Williams, *in press*).

Grazing by Canada Geese may yet become a factor on the Campbell River marshes, and is an example of the ‟moving target″ aspect of ecosystems discussed by Walters and Holling (1990). During the early years of this project, there was a small amount of grazing on the plug vegetation, but the birds did not seem to affect the growth of vegetation on the islands. Few Canada Geese used the estuary at that time. For example, during weekly bird surveys over the period 31 October 1982 through 18 March 1984, a total of only 31 Canada Geese was observed on or near the islands (Dawe et al. 1995). However, on the east coast of Vancouver Island, these introduced, resident geese have increased their populations significantly over the years, and roughly 250 or more geese can now regularly be found on the Campbell River estuary. Heavy browsing of the vegetation was noted on some of the islands in the estuary in 1994. On the Little Qualicum River estuary, some 95 km south of Campbell River, natural, monospecific stands of *Carex lyngbyei* have been eliminated through the browsing effects of large numbers of Canada Geese now using that system (N. K. Dawe, *unpublished data*).

The establishment of high-cover plant communities on the unplanted blocks suggests that transplanting may not be required, provided that there is a nearby seed or propagule source with suitable species, and provided that the substrate is stable and at suitable elevation. The proximity of Nunn′s Island and the Nunn′s Creek marshes probably had some influence on the success of marsh plant establishment on the unplanted blocks. Some of the colonization may also have resulted from seeds in the plugs of adjacent planted blocks or a seed bank in the dredge spoils. The results here suggest only a slight delay in success between planted and unplanted blocks. Frenkel and Morlan (1991) also note that planting may be unnecessary in some cases.

The concept of ecosystems as moving targets (Walters and Holling 1990) seems particularly well suited to the developing island marshes at Campbell River. Had our monitoring survey ended in 1988, we might have concluded that the project was a success, and that the planted marsh communities had achieved a species composition and cover similar to those of the natural marshes. Our 1994 results show, however, that the community composition of the islands is still changing. Also, the large areas devoid of vegetation on some of the planted blocks did not appear until some time after year 6, another example of slowly changing variables. A combination of island design and substrate settling that resulted in areas of standing water and waterlogging of the plants probably contributed to the dieback. This problem may have been avoided initially, had the islands been constructed with a rounded cross-section or been sloped like island 3 and Nunn′s Island.

Broome et al. (1986) found that a 10-yr sampling period was sufficient to compare the primary productivity of a transplant site with that of adjacent natural marshes, and to determine that the transplanted marsh was persistent and self-sustaining. Results of our study are not so clear, even after 13 years. Mitsch and Wilson (1996) advise a period of 15−20 years before judging the success of creating freshwater marshes, and they suggest that coastal wetlands may require even more time. Our study and others highlight the need for long-term monitoring of coastal marsh creation and rehabilitation projects (Race and Christie 1982, Zedler 1984, 1988, Frenkel and Morlan 1991, Dawe and McIntosh 1993). The time required will differ depending, in part, on the dynamic nature of the system, but should continue at least until annual variations in species composition, cover, and productivity of the planted vegetation are similar to those of a nearby natural system.

CONCLUSIONS

Because adaptive management is a process of "learning by doing," the following is a brief discussion of what we have learned from this project to date.

First, substrate stability is a prerequisite for any intertidal marsh creation project. There is little likelihood of success if the transplanted vegetation must deal with a shifting substrate.

Second, the created substrate should be of a type, and built to an elevation, as close to those of nearby intertidal marshes having the vegetation species composition and biomass levels that the project is trying to recreate. Because the substrate elevation affects the inundation periods and inundating water salinities, errors here could result in ultimate shifts in the species composition to a type not planned for, or could result in the species being eliminated altogether, as was the vegetation at the lower elevations of island 3 in this study.

Third, it seems likely that, if a stable substrate has been created at the right elevation and there is a proximate seed or propagule source, such as a nearby marsh, transplanting of plugs would not be necessary at all. This could be a significant factor for projects with few financial resources and lots of time.

Fourth, the system itself is a "moving target" evolving through natural and human impacts, and it has many potential outcomes (Walters and Holling 1990). The challenge here is to distinguish ongoing changes in the created system resulting from the project's management activities from the natural and human-influenced changes that would affect both the project and the natural system.

Fifth, many of the system variables change slowly on a temporal scale and can only be observed through long-term monitoring.

Sixth, this study emphasizes the importance of an adaptive approach to environmental projects, with long-term monitoring as a key component. With learning as an inherent objective (Johnson 1999), monitoring, evaluation, and response must take place (Walters 1986, Hilborn 1992). Monitoring, and then evaluating the results of the monitoring effort, allows managers to instigate new management actions or to modify the original management actions where necessary.

For example, as a result of our monitoring efforts in 1994, we were quickly able to react to the unforseen problems of the vegetation dieback due to ponding of the water on three of the islands. In an attempt to reverse the dieback, we created a number of channels to drain these areas, mimicking the dendritic channels of natural systems. The channels were dug on two of the islands, leaving one island with areas of dieback as a control. Another complete sampling session planned for 2002 will allow us to evaluate the effect of these new management actions. As Hilborn (1992) observes, "If you cannot respond to what you have learned, you really have not learned at all."

Finally, although this project does appear to be successful, at least based on the few factors of the system we have been monitoring, we repeat the cautions of Race and Christie (1982) and others regarding the use of marsh creation as mitigation for coastal development projects. The result may be the trading of natural coastal wetlands or mudflats for human-made marshes that may ultimately fail to become productive systems.

Fahselt (1988) notes that meeting conservation objectives through transplantation could suggest that habitat conservation of important natural areas is not urgent, and, as a result, development of these sites may be unwisely allowed, much as a "no net loss" policy may tend to do. Gibson et al. (1994) also "remain skeptical" that wetland creation can mitigate losses of natural wetlands; they emphasize the importance of preserving existing marshes.

Holling and Meffe (1996) argue against a reduction of the range of natural variation of any system. When such a reduction occurs, the system loses its resilience, becoming more prone to a change in structure and thus increasing its vulnerability to perturbations that normally could be buffered. Simenstad and Thom (1996) report that, after seven years of monitoring a total of 16 ecosystem attributes at their marsh creation site, only a few showed functional trajectory thresholds indicative of natural, mature systems. This suggests that created wetlands are much simpler systems, with less range in variation than natural wetlands, even after a number of years of apparently successful growth, and are likely to be more vulnerable to perturbations than are the natural systems.

As a management concept, Holling and Meffe (1996) suggest that system processes and variables should be maintained as a default condition rather than changing them, unless there is no other option. We, too, urge caution in trading existing, productive estuarine habitat for the promise of mitigating the impacts of developments through marsh creation techniques until we have a better, long-term understanding of the status of the existing marsh creation projects on the eastern Pacific coast.

RESPONSES TO THIS ARTICLE

Responses to this article are invited. If accepted for publication, your response will be hyperlinked to the article. To submit a comment, follow this link. To read comments already accepted, follow this link.

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APPENDIX 1

Percentage changes in mean cover (C) and frequency of occurrence (F) for vegetation treatments on the Campbell River estuary, British Columbia (1983–1994) (t = trace).

Species	1983	1984	1985	1986	1988	1994

	C	F	C	F	C	F	C	F	C	F	C	F
Nunn's Island low marsh (*n* = 20 relevés)												
Carex lyngbyei	36	85	41	100	49	85	60	90	51	95	46	95
Juncus balticus	12	30	19	35	12	25	11	25	11	35	21	55
Potentilla pacifica	8	10	10	15	10	15	26	10	6	15	8	10
Deschampsia caespitosa	2	10	t	5	3	10	6	20	2	15	15	10
Eleocharis palustris	7	40	18	45	16	35	17	55	20	55	1	15
Triglochin maritimum	6	15	20	10	26	15	23	20	15	15	15	5
Scirpus americanus	15	5	0	0	0	0	0	0	0	0	0	0
Scirpus validus	3	5	4	30	2	25	15	20	9	25	9	10
Lilaeopsis occidentalis	14	35	10	15	11	15	8	10	4	25	t	5

Species	1983		1984		1985		1986		1988		1994	
	C	F	C	F	C	F	C	F	C	F	C	F
Nunn's Island mid-to-high marsh (*n* = 18 relevés)												
Carex lyngbyei	7	47	11	94	10	100	14	94	11	82	15	78
Juncus balticus	24	71	35	94	21	94	26	94	23	88	39	100
Potentilla pacifica	9	88	17	100	21	94	25	100	25	94	17	89
Deschampsia caespitosa	4	35	4	65	16	71	19	82	8	88	12	72
Eleocharis palustris	12	24	10	71	21	65	14	65	10	65	1	39
Triglochin maritimum	1	24	6	65	6	71	6	59	3	65	3	89
Scirpus americanus	t	6	23	18	7	35	3	29	t	24	t	11
Lilaeopsis occidentalis	0	0	8	24	1	29	6	18	10	18	0	0
Castilleja miniata	4	35	3	6	t	6	15	6	t	6	0	0
Trifolium wormskjoldii	t	59	10	65	10	82	11	82	13	76	5	72
Sidalcea hendersonii	13	24	11	24	8	47	10	47	10	53	17	28
Agrostis alba	0	0	0	0	0	0	0	0	0	0	6	17
Hordeum brachyantherum	0	0	t	6	t	12	t	18	2	12	2	33
Aster subspicatus	0	0	8	53	6	59	5	65	6	65	7	61
Ranunculus cymbalaria	0	0	0	0	t	6	t	12	0	0	0	0
Plantago macrocarpa	5	24	10	24	14	65	15	65	6	76	12	61
Polygonum aviculare	t	12	0	0	0	0	0	0	0	0	0	0
Festuca rubra	0	0	t	24	5	41	4	35	6	47	15	50
Sium suave	0	0	2	12	2	12	2	18	2	18	4	50
Vicia americana	3	18	1	18	3	41	8	24	2	29	5	39
Daucus carota	0	0	1	29	2	29	10	18	1	41	t	17
Dodecatheon pulchellum	0	0	1	18	t	29	1	24	t	29	t	11
Mentha arvensis	0	0	0	0	0	0	0	0	0	0	t	6
Platanthera greenei	0	0	2	12	t	6	0	0	t	12	0	0
Sisyrynchium angustifolium	0	0	3	6	t	6	t	6	0	0	0	0
Hypericum formosum	0	0	9	12	2	24	6	18	11	24	0	0
Fritillaria camschatcensis	0	0	0	0	0	0	t	6	0	0	0	0
Ranunculus occidentalis	0	0	0	0	0	0	0	0	2	24	0	0
Poa pratense	8	12	0	0	0	0	0	0	0	0	0	0

Species	1983		1984		1985		1986		1988		1994	
	C	F	C	F	C	F	C	F	C	F	C	F
Planted blocks (*n* = 81 relevés)												
Carex lyngbyei	8	90	13	93	27	91	33	94	38	98	49	88
Juncus balticus	5	41	7	59	12	56	12	70	30	73	40	64
Potentilla pacifica	1	41	4	39	11	44	19	48	26	66	15	43
Deschampsia caespitosa	2	11	3	11	6	18	8	21	8	29	9	28
Eleocharis palustris	9	80	9	83	9	81	13	88	11	86	2	15
Triglochin maritimum	2	19	2	20	3	19	2	19	3	20	1	12
Scirpus americanus	2	16	7	18	5	14	3	16	2	13	t	5
Scirpus cernuus	2	21	t	21	5	23	3	24	t	25	3	1
Scirpus validus	1	43	1	51	2	34	2	35	t	14	t	1

Lilaeopsis occidentalis	2	35	7	44	16	49	14	76	12	74	t	1
Castilleja miniata	t	1	0	0	0	0	0	0	0	0	0	0
Trifolium wormskjoldii	2	6	15	3	11	4	12	8	12	6	0	0
Sidalcea hendersoni	t	3	0	0	t	3	t	1	0	0	0	0
Agrostis alba	t	3	2	4	t	1	t	1	0	0	0	0
Typha latifolia	t	1	0	0	0	0	0	0	0	0	0	0
Hordeum brachyantherum	t	1	0	0	0	0	0	0	t	1	0	0
Juncus bufonius	t	1	t	3	0	0	t	3	0	0	0	0
Ruppia maritima	t	1	0	0	0	0	0	0	0	0	0	0
Aster subspicatus	t	1	0	0	0	0	0	0	0	0	0	0
Ranunculus cymbalaria	0	0	0	0	t	1	t	5	0	0	0	0
Plantago macrocarpa	0	0	0	0	0	0	3	1	t	1	0	0
Plantago maritima	0	0	t	3	t	1	0	0	0	0	0	0
Polygonum sp.	t	1	t	1	0	0	0	0	0	0	0	0
Polygonum aviculare	t	1	0	0	0	0	0	0	0	0	0	0

Species	1983		1984		1985		1986		1988		1994	
	C	F	C	F	C	F	C	F	C	F	C	F
Unplanted blocks (*n* = 45 relevés)												
Carex lyngbyei	1	23	3	45	11	45	17	61	20	73	25	71
Juncus balticus	0	0	t	16	8	20	8	27	19	43	49	51
Potentilla pacifica	t	2	t	7	10	20	19	32	26	52	24	47
Deschampsia caespitosa	t	2	0	0	7	7	2	23	6	25	10	27
Eleocharis palustris	2	16	3	41	11	57	10	68	15	89	6	44
Triglochin maritimum	t	2	0	0	0	0	t	5	0	0	1	7
Scirpus americanus	0	0	t	2	0	0	3	2	0	0	0	0
Scirpus cernuus	1	16	2	30	8	48	5	52	5	41	7	16
Scirpus validus	t	50	2	55	7	34	7	20	10	25	13	7
Lilaeopsis occidentalis	t	2	2	9	7	18	10	36	22	48	3	2
Hordeum brachyantherum	0	0	0	0	t	2	3	2	3	2	0	0
Aster subspicatus	0	0	0	0	0	0	0	0	3	5	0	0
Ranunculus cymbalaria	0	0	0	0	t	5	0	0	0	0	0	0
Plantago macrocarpa	0	0	t	2	t	2	t	5	0	0	0	0
Ruppia maritima	t	2	0	0	0	0	0	0	t	2	0	0
Juncus bufonius	0	0	0	0	0	0	2	5	t	2	0	0
Polygonum sp.	0	0	0	0	0	0	t	2	0	0	0	0
Glaux maritima	0	0	0	0	0	0	0	0	t	2	0	0

Species	1983		1984		1985		1986		1988	
	C	F	C	F	C	F	C	F	C	F
Island 3, transects 10 and 11 (*n* = 11 relevés)										
Carex lyngbyei	1	36	2	45	6	55	8	45	11	45
Juncus balticus	0	0	3	9	26	18	3	9	45	45
Potentilla pacifica	0	0	2	18	14	36	19	64	18	82
Deschampsia caespitosa	t	9	t	27	3	45	13	73	9	64
Eleocharis palustris	0	0	8	18	t	9	3	9	3	9
Scirpus validus	t	36	2	18	t	9	t	9	0	0
Lilaeopsis occidentalis	0	0	0	0	2	18	t	9	3	9
Castilleja miniata	0	0	0	0	0	0	0	0	15	9
Trifolium wormskjoldii	0	0	t	9	5	27	5	36	17	45
Agrostis alba	0	0	t	18	2	73	4	45	2	27
Hordeum brachyantherum	0	0	t	9	0	0	0	0	t	18
Aster subspicatus	0	0	t	9	3	9	t	27	6	64
Plantago macrocarpa	0	0	0	0	9	18	0	0	0	0
Plantago lanceolata	0	0	2	18	t	45	5	55	11	73
Trifolium repens	0	0	0	0	3	9	t	55	32	64
Plantago major	1	27	9	45	19	73	8	73	1	64
Spergula arvense	2	36	6	73	0	0	t	18	13	55

Poa pratense	t	18	8	73	27	45	13	64	11	55
Spergularia rubra	28	64	39	91	39	91	19	91	6	55
Polygonum persicaria	6	73	4	82	1	36	1	45	t	27
Festuca rubra	0	0	0	0	0	0	0	0	2	18
Cirsium arvense	0	0	0	0	0	0	0	0	2	18
Lolium perenne	0	0	0	0	0	0	0	0	t	18
Rumex acetosella	0	0	0	0	t	18	0	0	t	18
Pyrus fusca	0	0	t	9	0	0	t	55	3	9
Matricaria matricarioides	0	0	0	0	0	0	0	0	t	9
Atropa belladona	0	0	0	0	0	0	t	9	0	0
Trifolium pratense	0	0	t	9	0	0	0	0	0	0
Chenopodium album	t	36	t	45	t	9	0	0	0	0
Epilobium watsonii	0	0	0	0	t	27	0	0	0	0
Sonchus arvense	0	0	t	27	0	0	t	55	0	0
Grindelia integrifolia	0	0	0	0	0	0	t	9	0	0
Hypochaeris radicata	0	0	t	9	0	0	t	9	0	0
Atriplex patula	0	0	0	0	0	0	t	9	0	0
Rosa nutkana	0	0	0	0	t	9	0	0	0	0

Address of Correspondent:
Neil K. Dawe
Canadian Wildlife Service, 3567 Island Highway West, Qualicum Beach, British Columbia, Canada V9K 2B7
Phone: (250) 752-9611
Neil.Dawe@ec.gc.ca