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FUNCTIONAL EQUIVALENCY TRAJECTORIES OF THE RESTORED GOG-LE-HI-TE ESTUARINE WETLAND^{1,2}

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Abstract. Assessing performance of restored and created wetlands for compensatory mitigation and restoration poses a mismatch between long-term processes and the short-term expediency of management decisions. If they were predictable, patterns in the temporal development of important wetland processes could reduce long-term uncertainty of the outcome of restoration projects. To test our ability to predict long-term trends and patterns in the development of a restored wetland based on the first 7 yr of its development, we analyzed 16 ecosystem functional attributes of the Gog-Le-Hi-Te Wetland, in the Puyallup River estuary, Puget Sound, Washington, USA. This estuarine wetland system was restored to tidal inundation in 1986. Only a few of the 16 ecosystem attributes analyzed showed functional trajectories toward equivalency with natural wetlands, and many were inconclusive or suggested disfunction relative to reference wetlands. Natural variability among reference sites also inhibited our ability to interpret an expected asymptote in developmental trajectories.

The ability of wetland managers to assess compensatory-mitigation success over short-term (e.g., regulatory) timeframes depends upon the selection of wetland attributes that can predict long-term trends in the development of the restored/created system. However, we are hampered by a basic lack of long-term data sets describing the patterns, trends, and variability in natural wetland responses to disturbance, as well as natural variability in wetland attributes in presumably mature wetland communities. Ultimately, it may be necessary to supplant our descriptive means of assessing functional equivalency with simple, controlled manipulative experiments or assays, standardized across restoration/mitigation and reference sites.

Key words: *fish and wildlife habitat; food web support; functional assessment; long-term equivalency; Puget Sound estuary; wetland restoration.*

INTRODUCTION

General goals of wetland restoration, creation, and enhancement are to reduce and ultimately reverse losses of wetland area and to reestablish natural hydrological, geochemical, and ecological processes that we have associated with various “functions” (e.g., improve water quality, protect shorelines, provide fish and wildlife habitat). The typical approach is to reproduce structural characteristics of natural wetlands under the inherent assumption that function will follow form. This presumes that scientists understand and can quantify fundamental wetland processes and the attributes that indicate wetland function, and that the patterns of development are predictable. Given the typically short (3–5 yr) life-span of monitoring programs instituted to

assess performance of restored or created wetlands, we need relevant predictors of system development and a better understanding of the time required to achieve functional equivalency with natural reference wetlands.

However: (1) relationships between wetland attributes and function are seldom predictable; (2) wetlands are not isolated, and thus large-scale hydrologic and other landscape processes must be considered when wetlands are manipulated; (3) few, if any, ecosystem processes are now uninfluenced by man, and the product of restoration may be entirely unrelated to the historic wetland conditions that are often the expected goal; and (4) the wetland “functions” we recognize are not necessarily representative of complex, random, disorganized, and evolutionary processes in nature (Katz 1991).

To develop an effective measure of “functional equivalency” of wetland restoration, creation, and enhancement activities, three basic questions need to be addressed: (1) How long does it take for constructed ecosystems to develop the functions of natural sys-

¹ Manuscript received 31 August 1994; revised 12 April 1995; accepted 13 April 1995; final version received 31 May 1995.

² For reprints of this group of papers on wetland mitigation, see footnote 2, page 33.

tems? (2) What factors limit the development of ecosystem functions in constructed wetlands? and (3) How might we mediate limiting factors and accelerate development? (Loucks et al. 1990). Wetland restoration lacks chronosequences >20 yr analogous to those used in forest reorganization (Reiners 1992).

Assessments of functional equivalency usually involve pair-wise comparisons between the restored, created, or enhanced wetland and a naturally occurring reference habitat. Such comparisons are characteristically short-term measurements of wetland structure. The use of short-term studies of structure, as opposed to long-term studies of processes, seriously limits our ability to predict how wetlands can be effectively restored and managed adaptively (Zedler 1988, National Research Council 1992). For example, although many studies find that some critical attributes of human-made salt marshes, such as the composition and density of macrofaunal assemblages (e.g., Moy and Levin 1991), resemble controls after as few as 3 yr, complex ecological interactions do not necessarily follow the same time frame. As a result, assessment usually does not occur over a time series sufficient to identify the functionally equivalent "endpoint" of community or ecosystem maturity (e.g., trophic linkages between benthic prey and fish utilizing the marsh; Moy and Levin 1991). Furthermore, high spatial and temporal variability in natural salt marshes, in conjunction with the limited intensity and number of reference marshes that can be feasibly sampled, limits the power of any statistical comparisons. Sufficient replication (as opposed to pseudoreplication, Hurlbert 1984) is rare. Furthermore, little time is allocated to measuring variability prior to the "treatment," which would allow statistical determination of the necessary number of replicates (Hairston 1989). Inherent differences between human-made wetlands and controls are especially problematic in heavily urbanized landscapes, where undisturbed wetlands are rare, small, and under stress. Model trajectories based upon a spectrum of constructed and natural wetland cases will be particularly important in urban and other stressed landscapes, because disturbance or stress levels may preclude functional equivalency (Richardson 1994). Such trajectories or performance curves (Kentula et al. 1992) may show indicators of wetland function (a) progressively approaching and then surpassing the mean (although often within temporal variation) for natural reference wetlands; (b) rapidly (linearly) converging to reference levels; (c) progressively developing by stabilizing at a level significantly lower than reference levels; (d) progressing very slowly before approaching reference levels, ultimately developing a sigmoid trend; or (e) starting at a higher level but slowly declining to below reference levels. If we knew whether, and which, wetland attributes would follow linear, asymptotic, logistic, or other development patterns, we could theoretically predict

the functional equivalency and contribute extensively to the future predictive power of wetland restoration.

To examine the viability of such functional equivalency trajectories, in this paper we review and summarize the structural and functional changes during the first 6 yr post-restoration of the Gog-Le-Hi-Te Wetland, a 3.9-ha brackish mitigation site in the Puyallup River estuary. Results of monitoring Gog-Le-Hi-Te have been reported in Thom et al. (1987, 1988, 1990, 1991) and Shreffler et al. (1990b); results of further experiments on juvenile salmon residence time and foraging appeared in Shreffler et al. (1990a, 1992). However, this is the first comprehensive synthesis of the Gog-Le-Hi-Te system since restoration of tidal inundation.

STUDY SITE

Gog-Le-Hi-Te is located near the upstream extent of salinity influence in the Puyallup River estuary (Fig. 1). From the historically broad expanse of 840 ha of mudflat and 1569 ha of estuarine emergent marsh, the Puyallup River delta has been extensively modified by industrial development that began in the 1920s. Since 1877–1894, 98.5% of the historical emergent marsh and 89.4% of the mudflat habitat have been lost (David Evans and Associates, 1991 *unpublished report* to U.S. Army Corps of Engineers Seattle District, Seattle, Washington, USA). Wetlands that remain are extremely small, fragmented and often contaminated with industrial and domestic wastes.

The Port of Tacoma constructed Gog-Le-Hi-Te in 1986 as mitigation for filling a comparably sized parcel of land containing both wetland and upland habitats. The Clean Water Act³ Section 404 permit allowing the fill specified two conditions: (1) the environmental impacts of the fill be mitigated through construction of a comparably sized or larger wetland; and (2) the ecological "performance" of the new wetland be monitored and the wetland be maintained in perpetuity. Restoration had a primary objective of providing fish and wildlife habitat, based on the area of restored wetland, as follows: juvenile salmon, 50% of the area; waterfowl, 20%; shorebirds, 10%; raptors, 10%; and small mammals, 10%. Accordingly, the restoration design incorporated 2.2 ha of estuarine wetlands, including intertidal sedge (*Carex lyngbyei*) and cattail (*Typha latifolia*) marshes (34% of intertidal area), unvegetated mudflats and tidal channels (66%), and 1.7 ha of upland grassland (76% of upland area), freshwater marsh (12%), and shrub and forested riparian habitats (12%) (Fig. 1).

Construction of the restoration site required excavation of $\approx 55\,000\text{ m}^3$ of fill material (i.e., dredged river, mudflat sediments, domestic solid waste) from behind the river dike, surface contouring, rerouting a gas pipeline, construction of a new flood control dike, and

³ Clean Water Act of 1977 and as amended, United States Code: Title 33: Section 1344.

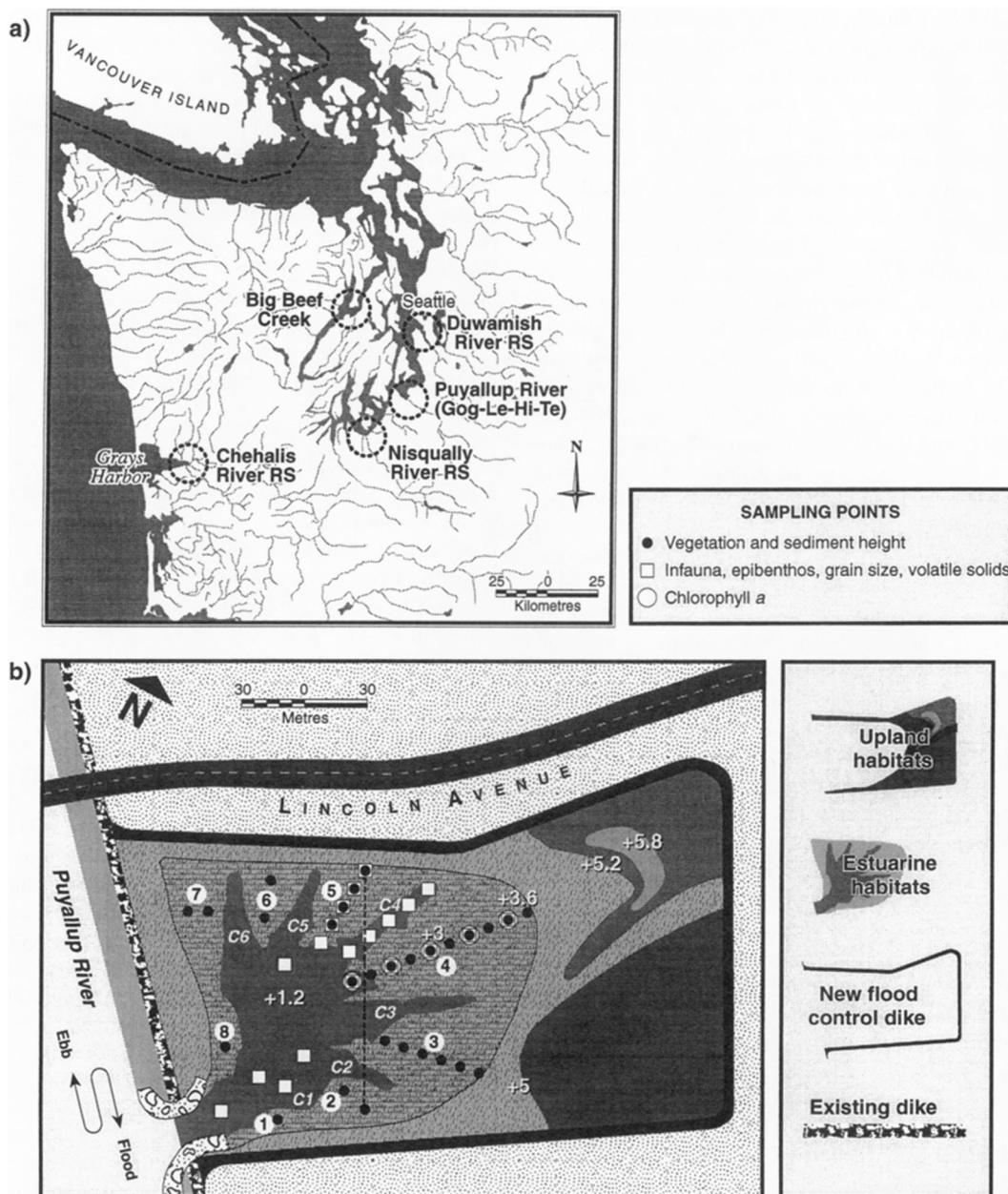


FIG. 1. Location (a) and design features (b) of the Gog-Le-Hi-Te restored wetland in the Puyallup River estuary, Puget Sound, Washington, USA. Locations of reference sites (RS) are designated by dashed circles on (a). In (b) sampling points and transects are indicated on intertidal flats numbered 1–8, tidal channels C1–C6, mid-basin (MB) and adjacent upland grassland, *Typha* marsh, and shrub-scrub freshwater wetland; dashed line indicates approximate location of the cross-sectional surface elevation profile illustrated in Fig. 5.

breaching of the river dike. Excavation and dike construction commenced in early July 1985. A pocket of polychlorinated biphenyls (PCBs) was discovered during excavation and had to be removed, delaying breaching of the dike and tidal inundation until February 1986. Culms of Lyngby's sedge, *Carex lyngbyei* Hornem., were planted onto unvegetated intertidal areas in March–July 1986 (37 159 culms) and April–May 1987

(11 650 culms). One of the seven contoured flats (Fig. 1b) was left unplanted to assess natural plant colonization and other attributes (e.g., benthic infauna and epibenthos).

Thom et al. (1987, 1990) describe normal water conditions in the wetland. Water column salinities are generally close to 0 ng/kg (0 ppt) between March and August. However, grab samples of channel waters in

the channels during late spring low tides have often found salinities >15 ng/kg, indicating salt intrusion and concentration (e.g., through evaporation) in the tidal channels. Nutrient concentrations were typical of freshwater streams tributary to Puget Sound, with high nitrogen-to-phosphorus ratios (N:P = 53.9 in March). The late summer data suggest that the available nitrogen was considerably less (N:P = 8.2). Ammonia concentrations were notably high (March, 6.34 ± 2.31 $\mu\text{mol/L}$; August, 4.91 ± 2.46 $\mu\text{mol/L}$) during both seasons.

Due to the lack of natural brackish emergent-marsh habitats remaining in Commencement Bay, reference sites for direct comparison to monitoring results from Gog-Le-Hi-Te were not available. However, comparable (i.e., same methodology, similar habitats) data from remnant mudflats (i.e., Hylebos Waterway) in Commencement Bay or from emergent marshes in other estuaries (i.e., Nisqually River and Duwamish River [Wenger 1995]) in Puget Sound or coastal Washington (i.e., Chehalis River estuary) were used when available (Fig. 1a).

METHODS

Fish and wildlife criteria and other measures of Gog-Le-Hi-Te's ecological performance were assessed by regularly monitoring: (1) topography, sediments, and vegetation; (2) water chemistry and temperature; (3) survival, distribution, and growth of the planted *Carex lyngbyei* and naturally recruited emergent plants; (4) benthic and planktonic invertebrate composition and standing stock; and (5) fish and bird species occurrence and density. Where possible, we sampled consistently for 7 yr; however, sediment accretion and limited monitoring resources often forced us to reduce the scope of our sampling. In addition, we conducted dedicated experiments to assess function directly; e.g., during 1987 and 1988 we appraised juvenile salmon (*Oncorhynchus* spp.) residence time, foraging, and growth because monitoring data on the fishes' occurrence and density were considered to be poor measures of wetland functioning (Shreffler et al. 1990a, 1992). More detailed descriptions of the associated methodologies and results are included in Thom et al. (1987, 1988, 1990, 1991) and Shreffler et al. (1990a, b, 1992).

Habitat and sediment structure

Annual aerial color photographs (22.9×22.9 cm color prints; 1:2400) were used to digitize (initially using a polar planimeter and, later, a microcomputer) surface areas of basic habitat polygons. Except for 1986, when the wetland was photographed in March, all photographs were taken during a spring low-tide series in July, at peak vegetative cover. Error in digitizing (differences between total area encompassed by digitizing and sum of separately digitized habitat polygons) ranged from 1 to 6.4%, with most errors $<3\%$.

Sediment grain size and organic content were mon-

itored at five points within each of the three basic habitat strata—mudflat/marsh, tidal channel, and basin—and sediment accretion was measured at 29 points over the eight intertidal flats (Fig. 1b). Sediments were sampled for grain size using a polyvinyl chloride (PVC) coring tube with 5.1 cm inside diameter inserted to 10 cm; smaller (2.1 cm diameter, 2 cm deep) cores for organic content were taken from within the larger cores. Grain size structure was determined by the dry-sieving technique of Folk (1968). They were washed in freshwater to solubilize salts and then oven-dried at 60°C before being mechanically shaken through nested 1.68-, 0.85-, 0.50-, 0.25-, 0.12-, and 0.06-mm sieves and the residual fines added to the original liquid fraction and processed by pipet analysis. Organic content was determined by percentage loss of mass of sample of sediment due to ashing at 500°C for 4 h in a muffle furnace.

Estimates of sediment accretion were based on short-term measurements of change in elevation relative to permanent markers, location of an artificial horizon, and long-term topographic surveys. One metre long wooden stakes were driven deep into the sediment at even intervals at 29 points along transects running from the base of the dike across the intertidal flats (Fig. 1b: Flats 1–8), with the top of the stakes ≈ 20 cm from the sediment surface. Repeated measurements were made of the distance from the top of the stakes to the sediment surface. Artificial horizons of plastic "glitter" flakes were established over five 100-cm² plots in May 1989. These plots were relocated in September 1989 and two 3 cm diameter cores were extracted from each. The cores were frozen, extruded from the PVC corer, and cut into 0.5-cm increments, and the sedimentation rate determined as the increment depth that contained the highest incidence of glitter flakes enumerated from the thawed increments under a dissecting microscope.

At the time of construction, and in November 1991, certified surveyors measured elevation over the entire wetland system and the dike, using a total station with vertical accuracy to 3 mm. These data were gridded and converted to surface contours using a microcomputer program, and differences in elevation integrated across the surface to estimate total accretion and erosion.

Primary producers

Sediment microalgae were assessed as chlorophyll *a* and phaeopigment concentration in sediment cores (1 cm in diameter \times 1 cm deep) taken at five positions (12, 28, 44, 60, and 76 m) along a 76 m long transect on Flat 4. Chlorophyll *a* and phaeopigment concentrations were determined by fluorometric analysis using ground sediment cores and 90% acetone extraction (Holm-Hansen et al. 1965).

Emergent marsh vegetation assemblages were sampled in various ways. In 1986 we recorded the frequency of occurrence for each plant species, as well

as bare substrate, at nested 10-cm intervals every 5 m along the 76-m transect line on Flat 4 (Fig. 1b). In 1987 and 1988 we estimated percentage cover from photographs taken directly above the center of each 1-m² plot and the percentage cover within the quadrat estimated by the point-intercept method using the projected 35-mm color slide. By 1989 *Typha latifolia* obscured other species and we used aerial photographs with field verification to estimate percentage cover. Shoot density was estimated using transects across each flat. Initial sampling in 1986 was from 5 m long and 0.5 m wide continuous-strip quadrats. Density data were obtained from 29 permanent 1-m² quadrats between 1987 and 1990; after 1990, shoot density was based on counts from three 0.1-m² quadrats placed haphazardly in the stands of each species on each intertidal flat. In 1990 an intensive survey of all angiosperm species was conducted along 54 transects across the intertidal portion of the wetland.

Aboveground biomass was estimated by subsampling shoots, measuring shoot length, weighing shoots after oven-drying at 90–100°C for 24–48 h, and developing a shoot length–dry mass relationship to estimate total dry mass. Belowground biomass was measured by extracting a 0.06-m², 30 cm deep core from the middle of each 0.1-m² quadrat, initially from all flats except Flat 5, and since 1990 on Flats 1, 2 and 7. The cores were washed over a 2-mm mesh sieve and live roots separated from dead material and detritus before being oven-dried and weighed.

Benthic and epibenthic invertebrates

Benthic macroinvertebrates were sampled (once or twice in most years, typically during late winter, early spring) before significant recruitment events. During the initial sampling year, we used a 2.75 cm inside-diameter core to sample the intertidal flats to a depth of 10 cm; in the channels and mid-basin we used a 0.05-m² square van Veen grab. In subsequent years we used a 5.1-cm core inserted to a depth of 10 cm to sample benthic infauna at 20 flat, channel, and mid-basin sites (Fig. 1b). Sediment cores were preserved in a 10% buffered formalin solution for 48 h, after which they were washed through a 0.5-mm mesh sieve and the retained organisms and other matter preserved in 70% alcohol. Organisms were identified under a dissecting microscope to major taxonomic levels (e.g., class) and enumerated.

We sampled epibenthic organisms at the same 20 sites using a suction pump that collects organisms from 0.018 m² of the benthic substrate. Samples were preserved in 10% buffered formalin. In the laboratory the

samples were stored in 50% alcohol, identified to species if feasible, and were enumerated. Wet mass was determined to the nearest 0.1 mg. Most of these samples were collected from March to June, when the maximum number of epibenthic-feeding juvenile salmon were expected.

Fish

Routine fish collections were made in tidal channels (predominantly channels 3 and 4) during low tide periods using a 9.7-m pole seine with 6-mm mesh bag. At each channel the seine was stretched across the width of the channel at the bayward end and then pulled the length of the channel to the upland end, where the net was drawn completely out of the water and the fish collected. Fish were sorted by species, counted, and released alive except for subsamples of juvenile salmon and other species of particular importance. In the laboratory they were measured and weighed and their stomach contents examined. Sampling that was more intensive and experimental was conducted in 1987 and 1988 to determine access to, and residence times and growth of, the most prominent juvenile salmon, chum (*Oncorhynchus keta*) and chinook (*O. tshawytscha*) (Shreffler et al. 1990a, 1992).

Birds

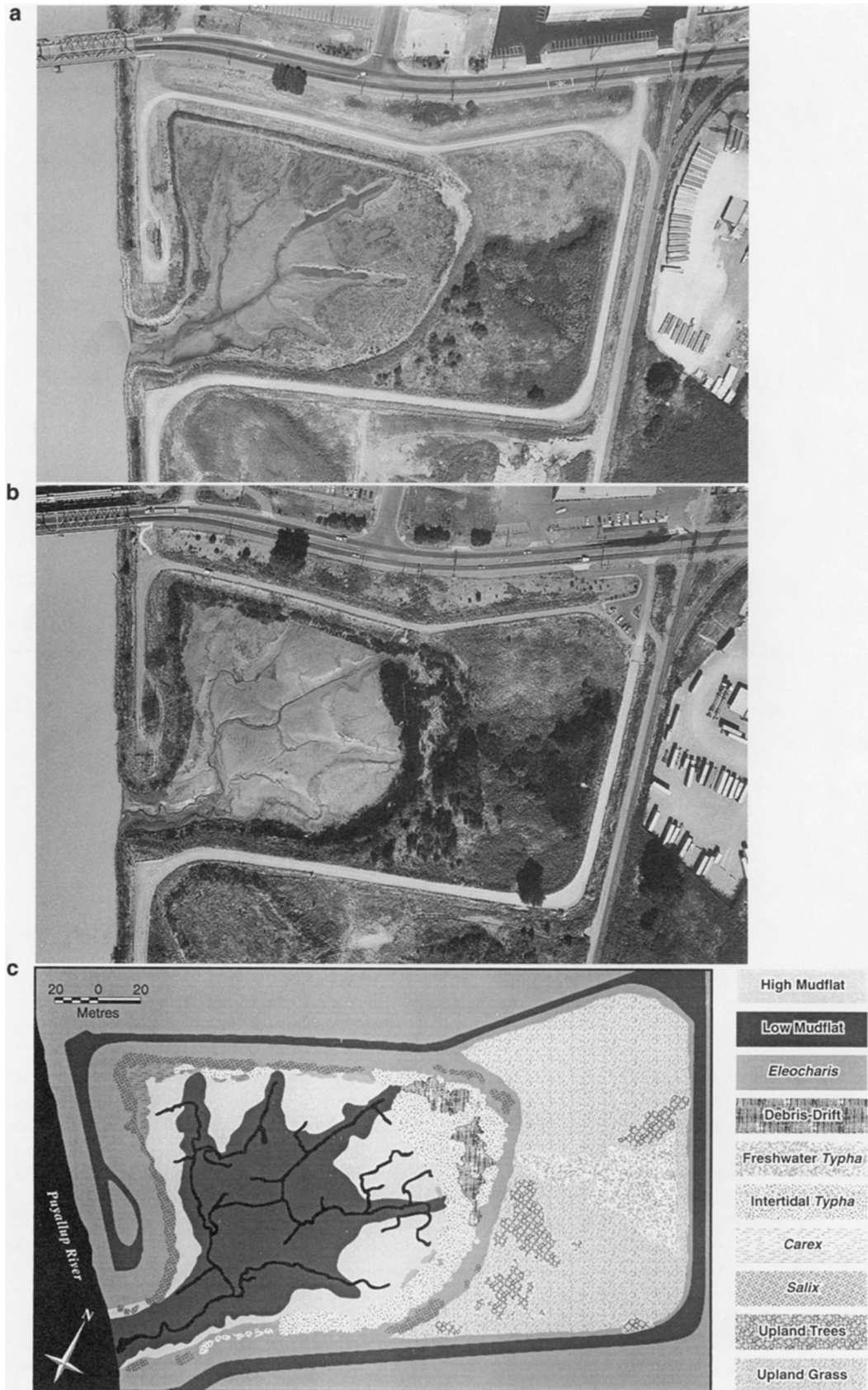
Birds were observed by members of the Tahoma Audubon Society during periodic (e.g., weekly to monthly) visits during the peak shorebird and waterfowl migration period, April–September. The abundance of each species was estimated and habitat occurrence recorded.

RESULTS

Habitat structure

Habitat structure of the nascent Gog-Le-Hi-Te wetland has been dynamic, including rapid retreat of planted *Carex lyngbyei* from lower intertidal elevations, expansion of naturally colonized, brackish emergent wetland species along upper edges of the intertidal flats, and sedimentation of the created tidal channels and basin (Fig. 2). Although the composition of upland habitats stabilized after the first 3 yr of modest areal expansion of willows (*Salix lasiandra*) and trees (*Alnus rubra*, *Rubus discolor*, *Populus trichocarpa*; Fig. 3a and b), intertidal habitats shifted during several episodes. A decrease in planted *C. lyngbyei* from the original ≈30% of the total intertidal area in 1987 to <2% in 1993 occurred predominantly between 1987–1988 and 1990–1993. Coincident with the decrease in *C. lyng-*

FIG. 2. Aerial photographs of Gog-Le-Hi-Te restored wetland in 1987 (a), one year after exposure to tidal inundation and planting of *Carex lyngbyei* sedge, and in 1993 (b); both photographs were taken during low spring tide series in July (see *Methods* and *Sediment habitat structure*). Photographs courtesy of Port of Tacoma. Habitat structure (c) of Gog-Le-Hi-Te restored wetland in 1993, as interpreted from aerial photographs and field verification.



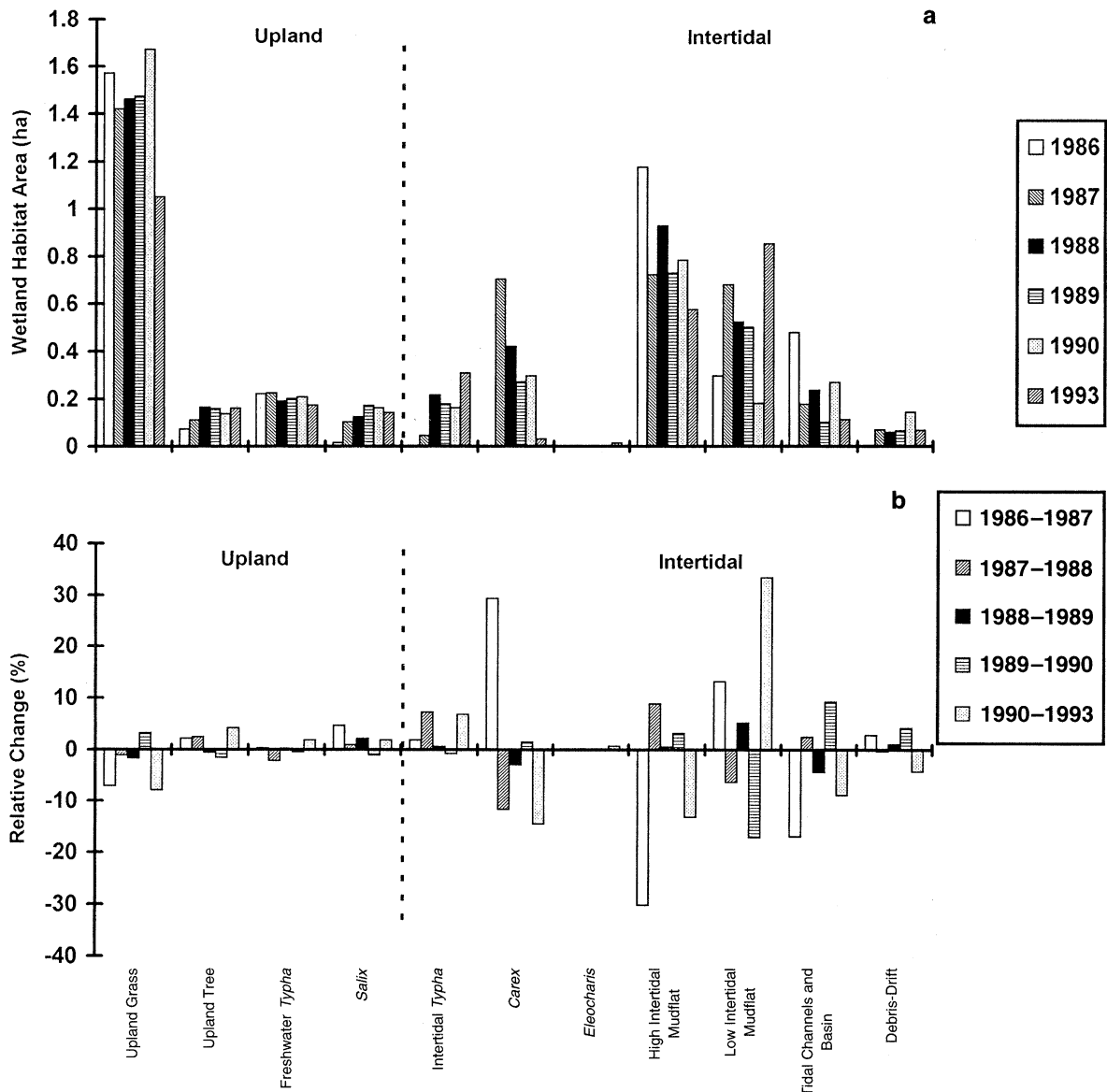


FIG. 3. Trends in habitat composition in Gog-Le-Hi-Te restored wetland between 1986 and 1993. (a) Absolute habitat area and (b) proportional change of upland and intertidal habitats.

byei were increases in intertidal *Typha latifolia* in both periods (Fig. 3b). *T. latifolia* advanced both out onto high intertidal mudflats—Flats 3 and 4—and around the intertidal perimeter of Flats 1 and 7. The only other natural colonizer that has formed a prominent habitat zone is *Eleocharis palustris*, which was prevalent at the upper margin of Flat 5 by 1990 (Thom et al. 1991; Fig. 2c). This was the only mudflat that was not planted with *C. lyngbyei*.

Although there was a consistent trend in decreasing tidal channel and basin area over the 7 yr and an overall increase in low intertidal mudflat (Fig. 3b), some variation is due to tide stage when aerial photographs were taken. Immediately after breaching of the dike, debris

(e.g., drift logs, tires) began accumulating in the northwest corner of the intertidal portion of the wetland (Fig. 2c), but the area occupied was relatively constant.

Sediment accretion has been extensive in both the tidal channels and basin and on the intertidal flats. Tidal channels, initially constructed 8–10 m wide and ≈ 1 m deep, immediately began decreasing in size, depth and sinuosity, while natural dendritic drainage channels began appearing on the mudflats in 1989. These increased in size and complexity (Fig. 4) and are now < 1 m wide and < 0.3 m deep. The drainage pattern has changed from essentially a first-order to a fourth-order system. Comparing cross-sectional profiles in August 1986 (“as-built”) and November 1990, indicates erosion of

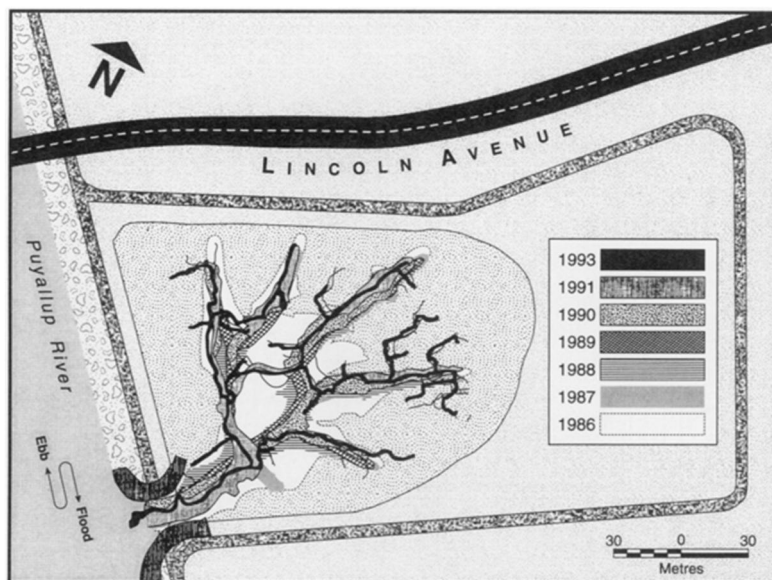


FIG. 4. Sequential changes in intertidal channel and basin configuration of Gog-Le-Hi-Te restored wetland between 1986 and 1993; lighter colors are oldest and darker colors youngest channel configurations.

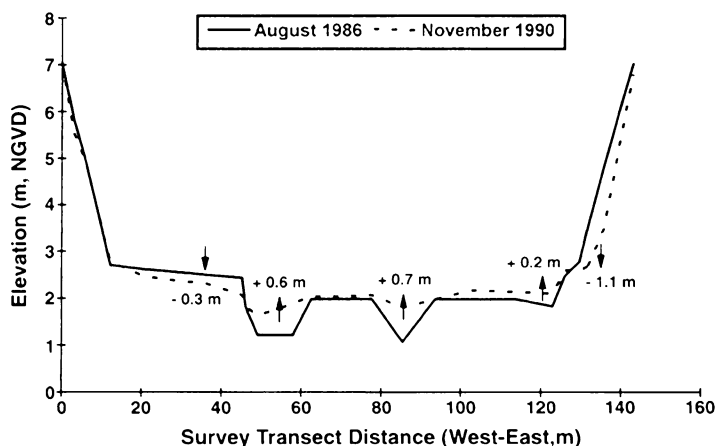
up to 1.1 m on the steep-gradient dike banks (both upland and high intertidal margins) and accretion of ≈ 0.2 – 0.3 m on the flats and up to ≈ 0.7 m in the tidal channels. Erosion was also evident on several portions of the intertidal flats (Fig. 5). Sediment accretion rates across Flats 1–7 between April 1986 and May 1989 was ≈ 1 – 3 cm/yr during the first 2 yr but up to 25 cm between 1988 and 1989. Erosion (≈ 1 – 4 cm/yr) was atypical (Shreffler et al. 1990b, Thom et al. 1990). Measurement at reference stakes re-established in 1990 indicated that sedimentation was insignificant during May–July (Thom et al. 1991). During the first 2 yr, accretion appeared constant at the upper elevations and outer edges of the intertidal flats, but increased dramatically over the intertidal flats in an accretion “event” between September 1988 and May 1989. Short-term (May–September) sedimentation rates dur-

ing 1989, derived from the artificial horizons, were estimated to be 0.4 cm/mo (Thom et al. 1990).

Total erosion and sedimentation and net deposition can be approximated by differences between surface plots of the 1986 and 1990 survey data. Gross erosion of 4101 m^3 occurred on the slopes of the dikes and margins of the intertidal areas, while the channels and flats experienced a gross accretion of 11495 m^3 , for a net accretion rate of 7394 m^3 over the 4.3 yr. Highest accretion occurred at the entrance channel of the wetland (Fig. 2b), where almost 2 m of sediment has accumulated.

Sediment composition tended to become coarser on the upper, landward portion of the intertidal flats and finer in the lower elevations adjacent to the basin. The percentage of fines (silt and clay; particle diameter $< 0.063 \text{ mm}$) decreased from an average of 88.2% to

FIG. 5. Differences in vertical elevation cross-sectional profile (heavy dashed line, Fig. 1b) of Gog-Le-Hi-Te restored wetland, indicating sediment accretion and erosion on intertidal flats and tidal channels, between August 1986 and November 1990 surveys; survey data courtesy of Port of Tacoma. Elevation is referenced to NGVD, United States National Geodetic Vertical Datum.



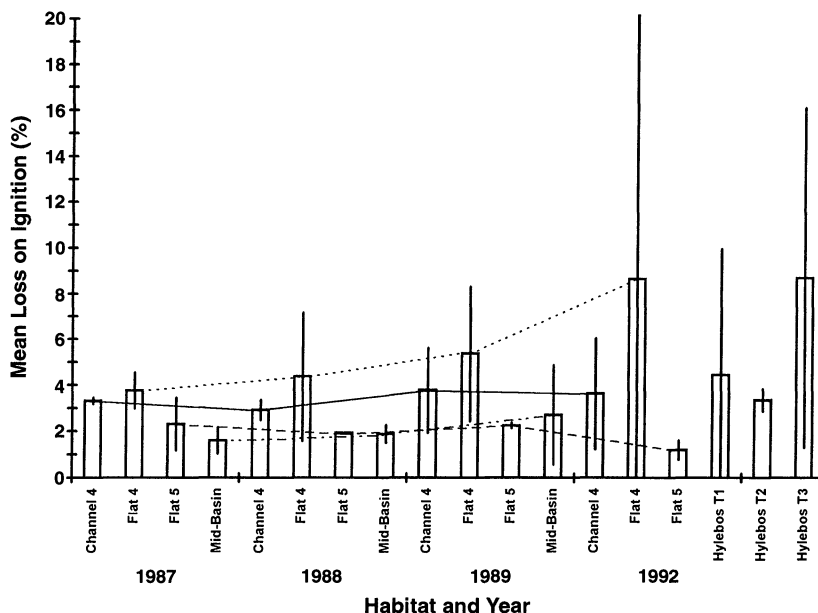


FIG. 6. Mean organic content (% loss on ignition) of sediments in Channel 4 (C4), Intertidal Mudflats 4 and 5 (F4, F5), Mid-Basin (MB) of Gog-Le-Hi-Te restored wetland, and three sites in Hylebos Waterway (HyT1-T3) between 1987 and 1992; see Fig. 3 for location of numbered channels and mudflats. Vertical lines on the bar indicate ± 1 standard deviation.

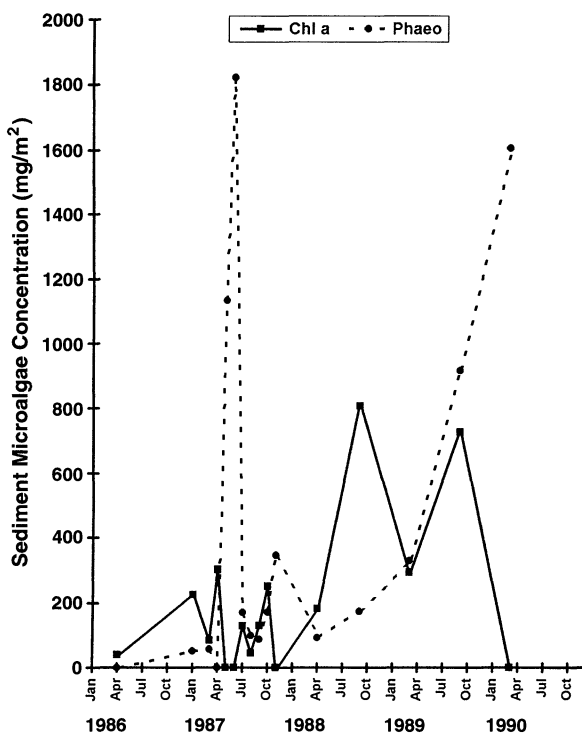


FIG. 7. Mean concentration of sediment microalgae biomass (chlorophyll *a* and phaeopigments) from Flat 4 of Gog-Le-Hi-Te restored wetland, 1986–1990.

80.2% at four of the five transect sites on Flat 4 and from 87.7% to 82.0% at two of the five transect sites on Flat 5 between 1987 and 1989. However, the percentage of fines at one of the mid-transect sampling sites on Flat 5 increased by 42.2% (from 51.1% to 93.3%). Over the same period, the percentage of fines increased between 72.1% and 82.7% on the outer, lower elevation sites. Increase in percentage of fines in the channels and basins was immediate and ubiquitous. Between 1987 and 1988, the percentage fines increased an average of 25.1% in Channel 4 and 47.6% in the Basin; over the 2 yr the average increase was 37.3% (from 59.2% to 96.5% total fines) in the channel and 65.7% (from 5.8% to 85.9%) in the Basin.

Organic content of the sediments in the unvegetated habitats of Channel 4, Flat 5 and the Basin remained relatively constant at 2–4% loss on ignition between 1987 and 1992, but appeared to increase on Flat 4 (Fig. 6) despite the disappearance of the *C. lyngbyei* over that period. Levels from three comparable “reference” mudflats in another intertidal area of Commencement Bay (Hylebos Waterway) in 1992 ranged between 3.3% and 8.7% loss on ignition.

Primary producers

As indicated by sediment chlorophyll *a*, benthic microalgae were immediately present on the intertidal mudflats. Mean concentrations during representative months (e.g., March–May, September) suggested a gradual increase between 1986 and 1990, although temporal variation was high (e.g., coefficient of variation $>100\%$) and chlorophyll *a* was often undetectable even in later years (Fig. 7). Phaeopigments indicated deg-

TABLE 1. Number of emergent plant species present on different intertidal flats of Gog-Le-Hi-Te wetland (Puget Sound, Washington, USA) during initial restoration of tidal inundation (1986) and four years later (1990). See Fig. 1 for location of specific flats in wetland. The one species in 1986 was the transplanted sedge *Carex lyngbyei*.

Flat No.	Flat area (m ²)	Number of emergent plant species		No. of taxa added/yr
		1986	1990	
1	210	1	17	4
2	750	1	26	6.2
3	2575	1	33	8
4	4250	1	40	10
5	1575	0	29	7.2
6	920	1	22	5.2
7	1350	1	7	1.5
8	275	1	6	1.2

radiation products and were often high after 1986, and especially so following an April 1987 bloom of sediment microalgae.

By 1990, 56 vascular plant species had colonized the intertidal portions of the wetland. Prominent taxa included *Typha latifolia*, a spikerush (*Eleocharis palustris*), and several sedges (*Carex stipata*, *C. deweyana*, and *C. lyngbyei*), rushes (*Juncus balticus*, *J. effusus*, *J. accuminatus*, *J. bufonius*), bulrushes (*Scirpus acutus*, *S. cyperinus*), and willows (*Salix lasiandra*, *S. hookeriana*, *S. scouleriana*) (Thom et al. 1991). Almost all were concentrated along the higher margins of the intertidal flats, although there was a consistent increase in low-growing and juvenile flowering plants, dominated by immature *J. bufonius*, at the seaward edge of the *C. lyngbyei* and *T. latifolia*. The total area of the intertidal flats corresponded directly with the number of species that had colonized the flats by 1990 (Table 1).

Carex lyngbyei production attributes were monitored intensively for comparison to reference data from natural *C. lyngbyei* marshes in other estuaries that we have sampled since 1986. Shoot density followed a gradual,

essentially linear increase from the time of transplantation until 1992, when it reached a maximum of 486 shoots/m², before declining precipitously to <5 shoots/m² with the overall decline in the remaining *C. lyngbyei* habitat by 1993 (Fig. 8). Shoot density in Gog-Le-Hi-Te achieved reference levels at the primary source population (Big Beef Creek, 1986), the Chehalis River estuary (1991–1993) and the Nisqually River estuary (1993) within 3 yr, except for the extremely high densities (mean = 905 shoots/m²) found in the Duwamish River estuary in 1993 (Wenger 1995).

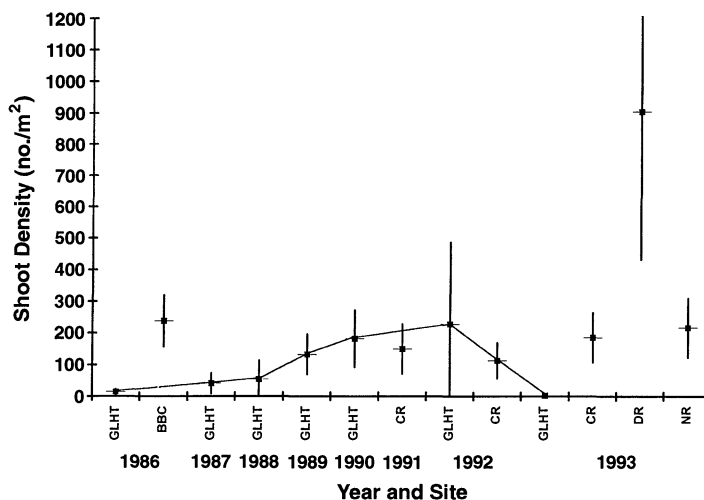
Aboveground biomass of *Carex lyngbyei* was sampled less frequently. The maximum biomass was well below the mean for reference sites (Fig. 9). One patch remaining in 1993 had an aboveground dry biomass of 827 g/m². The two reference marshes (Big Beef Creek and the Chehalis River estuary) had high biomass, although there was considerable variability in these estimates. The Duwamish River and Nisqually River estuaries (85–300 g/m²) were lower in aboveground dry biomass than even the transplanted *C. lyngbyei* in 1986 (Fig. 9; Wenger 1995).

Belowground dry biomass of *C. lyngbyei* was never high (Fig. 10). Values from reference marshes were highly variable, ranging from 198 to 8844 g/m² among years at the same sites sampled in the Chehalis River estuary, while extensive sampling at the Duwamish and Nisqually rivers in 1993 (Wenger 1995) provided means of 1107 and 443 g/m², respectively.

Benthic and epibenthic invertebrates

Benthic infaunal invertebrates immediately colonized the channels and basin and their densities continued to increase to >1500 individuals/100 cm² by 1989, but development on Flat 4 was slow (Fig. 11). Oligochaetes, nematodes and insect larvae (Chironomidae) predominated the initial colonization, but harpacticoid copepods (e.g., *Huntelemania jadensis*, *Microarthridion littorale*) and gammarid amphipods (e.g.,

FIG. 8. Mean shoot density of *Carex lyngbyei* in the primary growth zone of the sedge habitat between 1986 and 1993 at the Gog-Le-Hi-Te restored wetland (GLHT), and reference sites (BBC = Big Beef Creek, Hood Canal; CR = Chehalis River estuary, coastal Washington; DR = Duwamish River estuary, Puget Sound; NR = Nisqually River estuary, Puget Sound). No data were available from Gog-Le-Hi-Te in 1991. Vertical bars indicate ± 1 standard deviation.



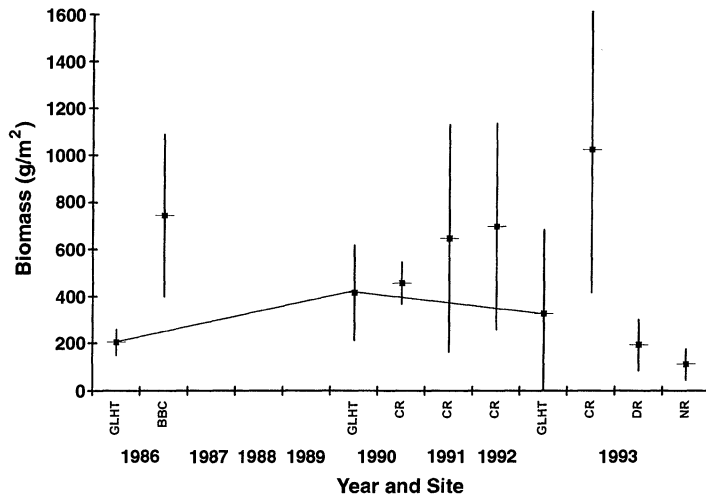


FIG. 9. Aboveground mean dry biomass of *Carex lyngbyei* in the primary growth zone of the sedge habitat between 1986 and 1993 at the Gog-Le-Hi-Te restored wetland. Abbreviations as in Fig. 8; vertical bars indicate ± 1 standard deviation.

Corophium spinicorne, *Eogammarus confervicolus*) appeared in 1987. Harpacticoid copepods ultimately superseded oligochaetes in both numerical (Fig. 12a) and gravimetric (Fig. 12b) importance within 3 yr. The taxa richness of harpacticoid copepods, the only taxa we sorted specifically, increased progressively, with the biggest increase in the 1st yr (Fig. 12c). *Byrocampa* sp. and *Maraenobiotus* sp. were the first epibenthic harpacticoid copepods to appear; by 1987, *Microarthridion littorale*, *Tachidiu discipes*, *T. triangularis*, *Nitocra spinipes armata*, *Huntmannia jadensis*, *Leimia vaga*, and *Mesochra lilljeborgi* and *M. rapiens* were identified; *Onchycampa mohammed* and *Nannopus palustris* appeared in 1988; and *Pseudobryadia* sp. were added in 1989.

Fish

Fishes immediately occupied the intertidal habitat. During the March-June sampling window (coincident with juvenile salmon migration), we captured 11 species in 1986, 16 species in 1987, and 9–19 species

between 1988 and 1990. The average number of species, standardized by days of sampling effort, was relatively constant at 11–12 for the beach seine samples irrespective of sampling effort (3–9 seines), but declined to 9 species by 1990. This increased (15–19 species) slightly with the intensive fyke net sampling in 1987 (40 d) and 1988 (110 d) (Shreffler et al. 1990a, 1992). No new species appeared after 1989 (Thom et al. 1991). The density and standing stock of fish utilizing the channels and basin increased slowly (Fig. 13). Densities were consistently <0.1 fish/m² from 1986 through 1988, but were almost 0.7 fish/m² in 1990. Species such as longnose dace (*Rhinichthys cataractae*), reddsideshiner (*Richardsonius balteatus*), threespine stickleback (*Gasterosteus aculeatus*), prickly sculpin (*Cottus asper*), Pacific staghorn sculpin (*Leptocottus armatus*), and starry flounder (*Platichthys stellatus*) were consistent residents in the wetland, generally increasing through 1989 (Table 2). Even the transient species, such as the juvenile salmon (chum salmon, *Oncorhynchus keta*; coho salmon, *O. kisutch*;

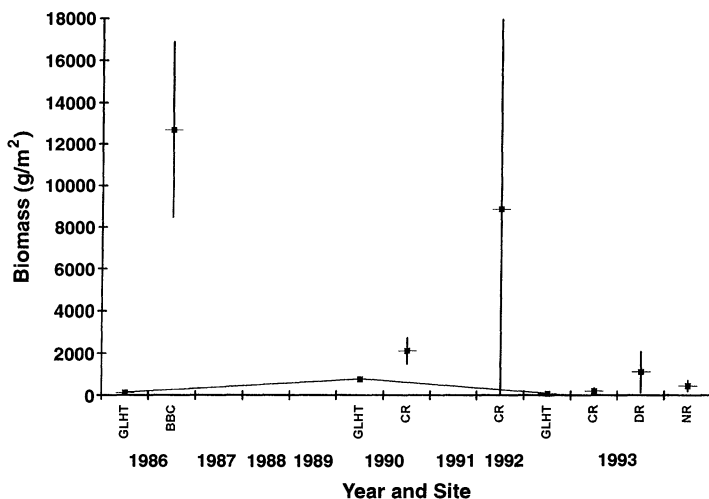


FIG. 10. Belowground mean dry biomass of *Carex lyngbyei* in the primary growth zone of the sedge habitat between 1986 and 1993 at the Gog-Le-Hi-Te restored wetland. Abbreviations as in Fig. 8; vertical bars indicate ± 1 standard deviation.

chinook salmon, *O. tshawytscha*) appeared to increase through 1989, suggesting that critical habitat functions such as prey resources and refuges from predation were developing. The more intensive sampling and experimentation in 1987 and 1988 (Shreffler et al. 1990a, 1992) indicated that residence time (1–9 d for chum, 1–43 d for chinook), prey composition (chironomid larvae and pupae, plecopterans, and adult dipterans), and growth (5.2 mg/d for chinook) of juvenile coho and chinook salmon were comparable to that in the meager literature from natural estuarine systems.

Birds

A diverse assemblage of birds immediately occupied Gog-Le-Hi-Te. Almost 70% (80) of the 112 species documented by 1990 were observed in the wetland in the 1st yr of its development (Fig. 14). Between 100 and 500 individuals were counted on any observation day, and densities were usually highest during the spring and fall migration periods. Initially, in 1986–1987, there was little consistent distribution of individuals in particular habitats, although the riparian border, cattail, and intertidal flats frequently supported the highest abundances (Fig. 15a). However, by 1990 the open water, intertidal flats, and dike consistently supported more birds than the border, cattail, aerial and upland habitats. In the 1st yr the various assemblages occurred equally (Fig. 15b). Thereafter, waterfowl dominated (40–85%) except during the summer, when other birds were comparable. Shorebirds seldom constituted >10% of the total individuals until August,

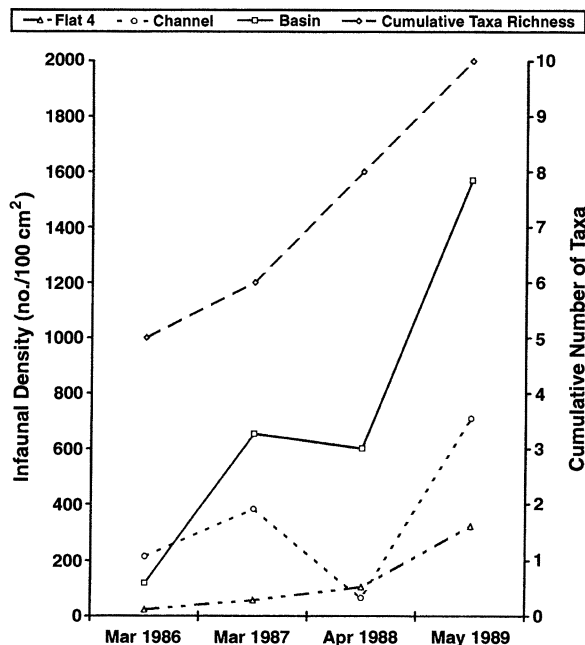


FIG. 11. Densities and cumulative taxon richness of benthic infauna in three habitats (intertidal flat, channel, and mid-basin) at the Gog-Le-Hi-Te restored wetland, 1986–1990.

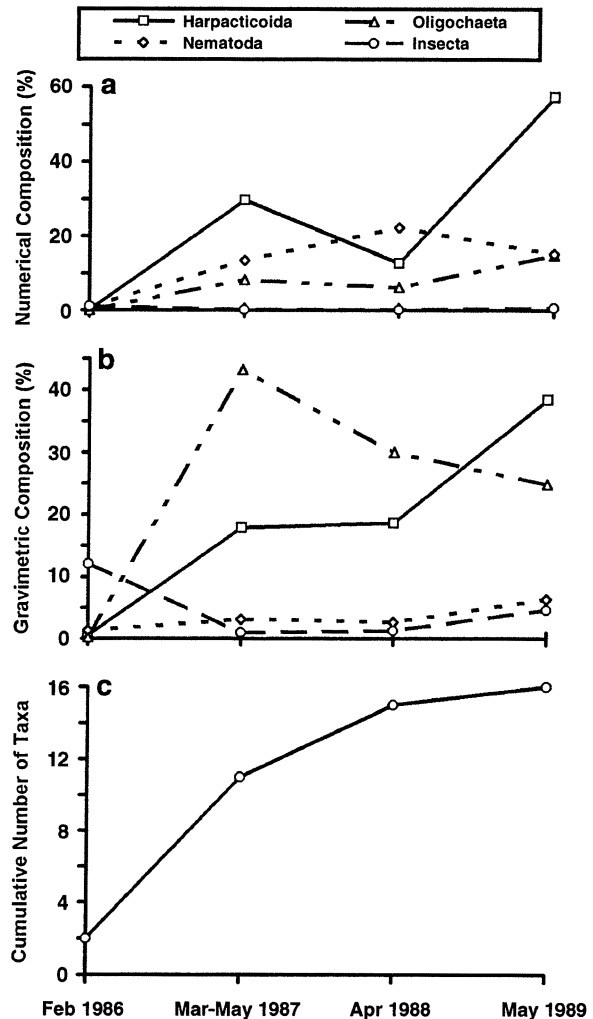


FIG. 12. Relative numerical (a) and gravimetric (b) composition, and harpacticoid copepod cumulative taxa richness (c), of epibenthic organisms at the Gog-Le-Hi-Te restored wetland, 1986–1989.

when they approached or exceeded waterfowl abundances.

DISCUSSION

The 7-yr monitoring history of Gog-Le-Hi-Te indicates few predictable trajectories of community development and system function. The time series data show various trajectories, only a few of which illustrate thresholds indicative of system maturity. Taxa richness of epibenthic organisms and fishes, and densities of fishes, all approached asymptotic trajectories within 3–5 yr, and three indicators of bird usage approximated quadratic trajectory forms, indicating extremely rapid responses. Alone, these indicators would contradict arguments that restoration, creation, and enhancement of estuarine marshes is problematic and proceeds by trial and error (Zedler 1988, Moy and Levin 1991). However, other ecological functions indicate that Gog-Le-

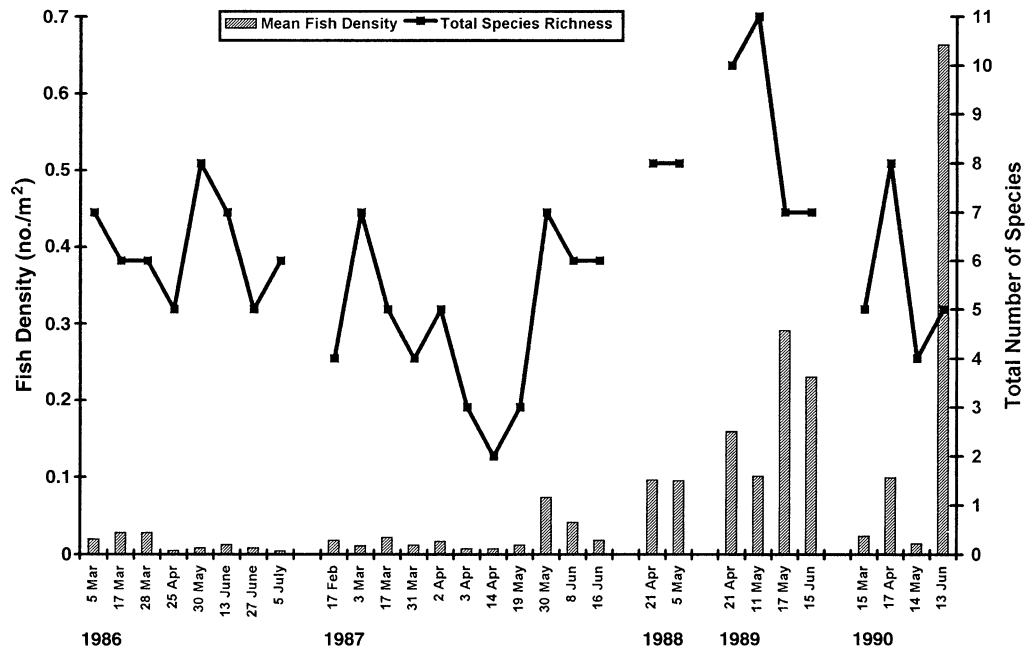


FIG. 13. Mean density and total species richness (no. of species captured) of fishes in tidal channels at the Gog-Le-Hi-Te restored wetland, 1986–1990.

Hi-Te is still in an early stage of development or on a pathway separate from comparable brackish wetland communities in this region. Sediment organic content, chlorophyll/phaeophytin pigments, infauna taxa richness and density, and proportion of harpacticoids and oligochaetes in the epibenthos increased slowly or re-

mained relatively depressed over the same 3–5 yr monitoring period. Furthermore, the various measures of *Carex lyngbyei* production illustrated gradual progression toward reference site levels, only to decline in year 7.

A longer time series is needed to determine if these

TABLE 2. Mean density (no. of fish/100 m²) of most common fish species in tidal channels at the Gog-Le-Hi-Te restored wetland, 1986–1990.

Species	1986								1987		
	5 Mar	17 Mar	28 Mar	25 Apr	30 May	13 Jun	27 Jun	5 Jul	17 Feb	3 Mar	17 Mar
Mountain whitefish, <i>Prosopium williamsoni</i> (Girard)	0.38	0.40	...
Pink salmon, <i>Oncorhynchus gorbuscha</i> (Walbaum)
Chum salmon, <i>Oncorhynchus keta</i> (Walbaum)	0.04	0.09	...	0.10	0.03	...	0.11	0.01
Coho salmon, <i>Oncorhynchus kisutch</i> (Walbaum)	0.13	0.21	0.01	0.03
Chinook salmon, <i>Oncorhynchus tshawytscha</i> (Walbaum)	0.04	0.01	0.16	0.36	0.04	0.20	0.60
Longnose dace, <i>Rhinichthys ataractae</i> (Valenciennes)	0.10	0.10	0.60
Redside shiner, <i>Richardsonius balteatus</i> (Richardson)	0.03	0.13	0.50	0.10	0.20
Largescale sucker, <i>Catostomus macrocheilus</i> (Girard)	1.15	0.68	0.68	0.01	...	0.03	...	0.10	0.20
Threespine stickleback, <i>Gasterosteus aculeatus</i> (Linneaus)	...	0.51	0.51	0.14	0.15	0.25	0.07	0.10	...	0.10	...
Prickly sculpin, <i>Cottus asper</i> (Richardson)	0.13	0.83	0.83	...	0.03	0.19	0.36	0.14
Pacific staghorn sculpin, <i>Leptocottus armatus</i> (Girard)	0.03	0.05	...	0.01	0.80	0.10	0.40
Starry flounder, <i>Platichthys stellatus</i> (Pallus)	0.09	0.17	0.17	...	0.16	0.40	0.26

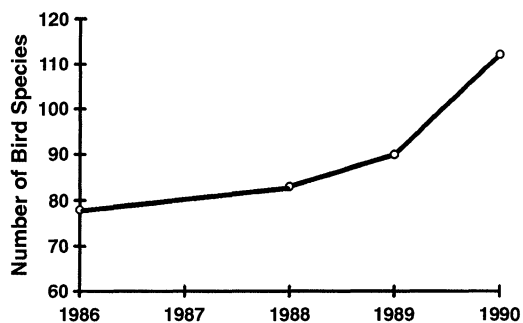


FIG. 14. Number of bird species observed at the Gog-Le-Hi-Te restored wetland, 1986–1990.

trajectories represent a certain stage of brackish tidal community development, comparable to the post-disturbance response to the resetting of community succession (Sousa 1979, Sutherland 1990). Unlike the rocky intertidal community, whose responses to disturbance have been described and modeled (Paine and Levin 1981), there are no models of brackish marsh patterns, processes, and rates of development after disturbance. Monitoring of *Carex lyngbyei* (Figs. 8–10) in the high intertidal mudflats, irrespective of distribution changes (Fig. 5a and b), would have predicted development to reference levels within 3–5 yr, but would not have foretold the rapid decline in standing stock associated with the dramatic retreat and virtual disappearance of *C. lyngbyei* as a prominent intertidal habitat. The slow development of belowground biomass over that period may have been a critical indicator

of the persistence and resilience of the planted *C. lyngbyei* stand to a physical or physiological disturbance. We can partially explain the collapse of the *C. lyngbyei*: the tidal elevation of the recontoured wetland was at the lower margin of the species' vertical range; due to freshwater tidal to oligohaline conditions, *Typha latifolia* was able to establish and overgrow the transplanted *C. lyngbyei* at the upper levels of its distribution; and other natural disturbances (e.g., intensive Canada Geese grazing, extremely low river flows, and increased salinity intrusion) may have stressed the *C. lyngbyei* beyond its tolerance limits. The majority of these explanations suggest extremely tight coupling among estuarine hydrogeomorphology, soil development, physicochemical and biological disturbance, and plant physiology and survival, all of which tend to be unappreciated or unevaluated in most wetland restoration monitoring.

Most monitoring time periods for mitigation and restoration projects are not sufficient to encompass natural perturbations that would test persistence and resilience of the developing community, nor are monitoring attributes sufficient to predict resistance to natural or human-made disturbances or recovery rates thereafter (Zedler 1987). It may be impossible to predict the outcome from even a 5–10 yr time series. Brown-Peterson (1993) found that, while newly recolonized (<4 yr) seagrass meadows had fish assemblages comparable to those in adjacent natural meadows, created meadows that were well established (31 yr) were inherently different from natural meadows. Streever and Crisman

TABLE 2. Continued.

1987								1988		1989				1990			
31 Mar	2 Apr	3 Apr	14 Apr	19 May	30 May	8 Jun	16 Jun	21 Apr	5 May	12 Apr	11 May	17 May	15 Jun	15 Mar	17 Apr	14 Mar	13 Jun
...
...	1.80
0.10	0.40	0.10	0.70	2.30	1.50	2.00	0.20	0.10	0.10
...	6.50	0.05	0.60	...	0.42	0.07
0.60	0.50	...	0.60	0.70	3.30	3.10	0.10	1.27	5.60	0.40	1.50	2.37	0.56	...	56.20
...	0.40	0.20	0.20	...	0.10	0.06	1.44
...	0.03	0.13	0.50	0.50	0.04	...	0.05
...	0.10	0.20	0.60	0.10	0.40	2.10	1.00	1.30	2.70	...	6.10	0.03	6.01	0.57	2.47
...	0.30	1.30	0.30	0.70	0.20	0.30	5.10	4.30	2.30	10.60	0.42	1.66	0.76	3.90
0.10	0.20	0.10	0.20	...	0.10	...	0.08	0.10	0.10	0.40	0.06	0.03	2.25
0.40	0.50	0.50	0.10	...	0.10	0.40	0.40	1.40	0.60	0.10	0.60	1.50	3.00	1.44	0.07	0.03	...
...	0.10	0.10	0.05	...	0.10	0.30	0.50	3.00	...	0.04	...	1.47

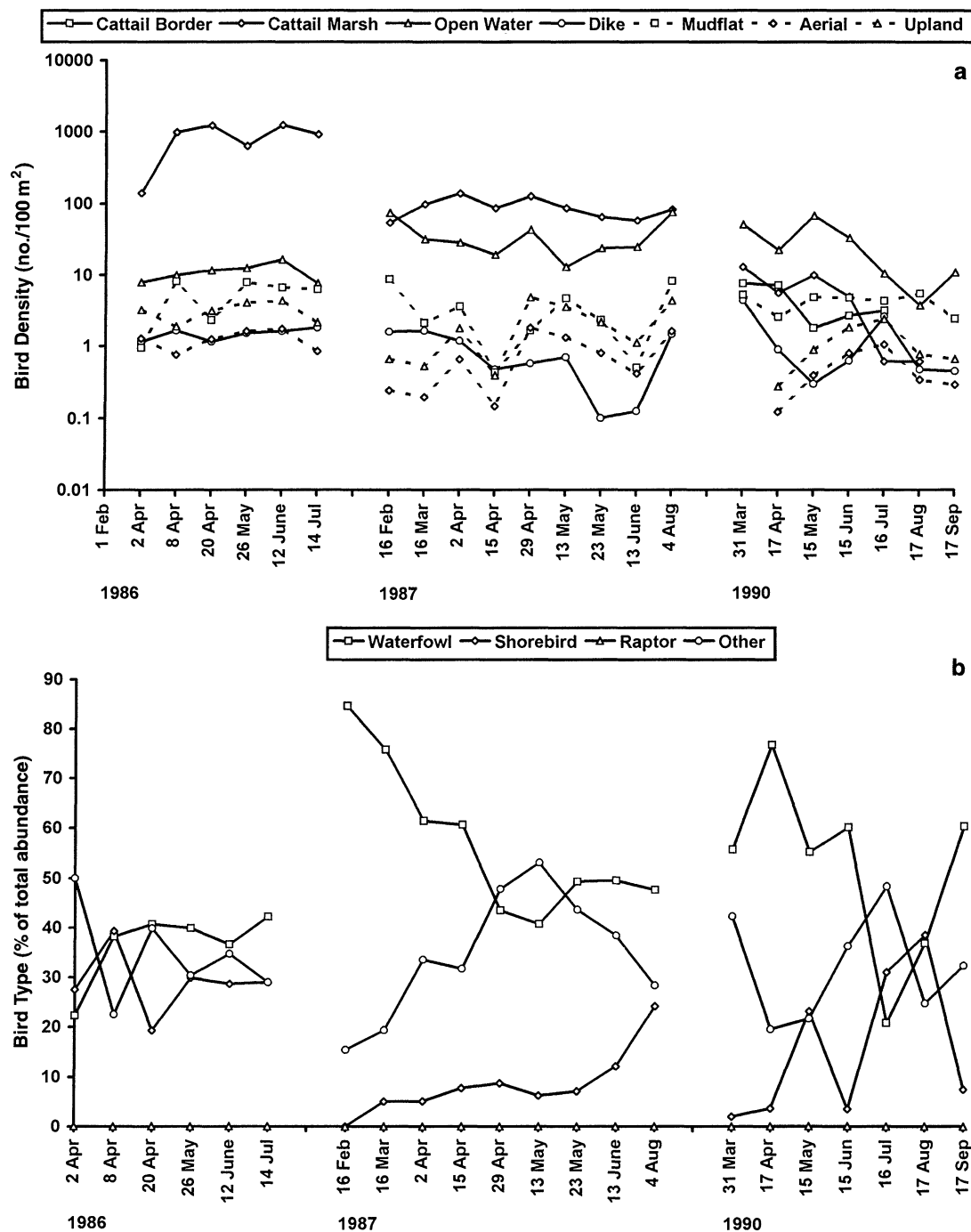


FIG. 15. Densities of birds observed in seven habitats (a) and relative composition (b; % of total abundance) by assemblage of birds at the Gog-Le-Hi-Te restored wetland, 1986–1990.

(1993) also showed that meiobenthic cladoceran assemblages and species richness in constructed wetlands up to 10 yr old were dissimilar from natural wetlands, which had greater variability. Whether the endpoints in these examples differ as a result of lagged, interrupted, or regressive trajectories, differences in local environmental conditions regulating community struc-

ture, or alternative states of mature communities (Sutherland 1990), cannot be determined without longer time-series.

Monitoring often focuses on structural attributes rather than the dynamic processes that determine a restored or constructed wetland's development, e.g., independent variables such as hydrology, sedimentology,

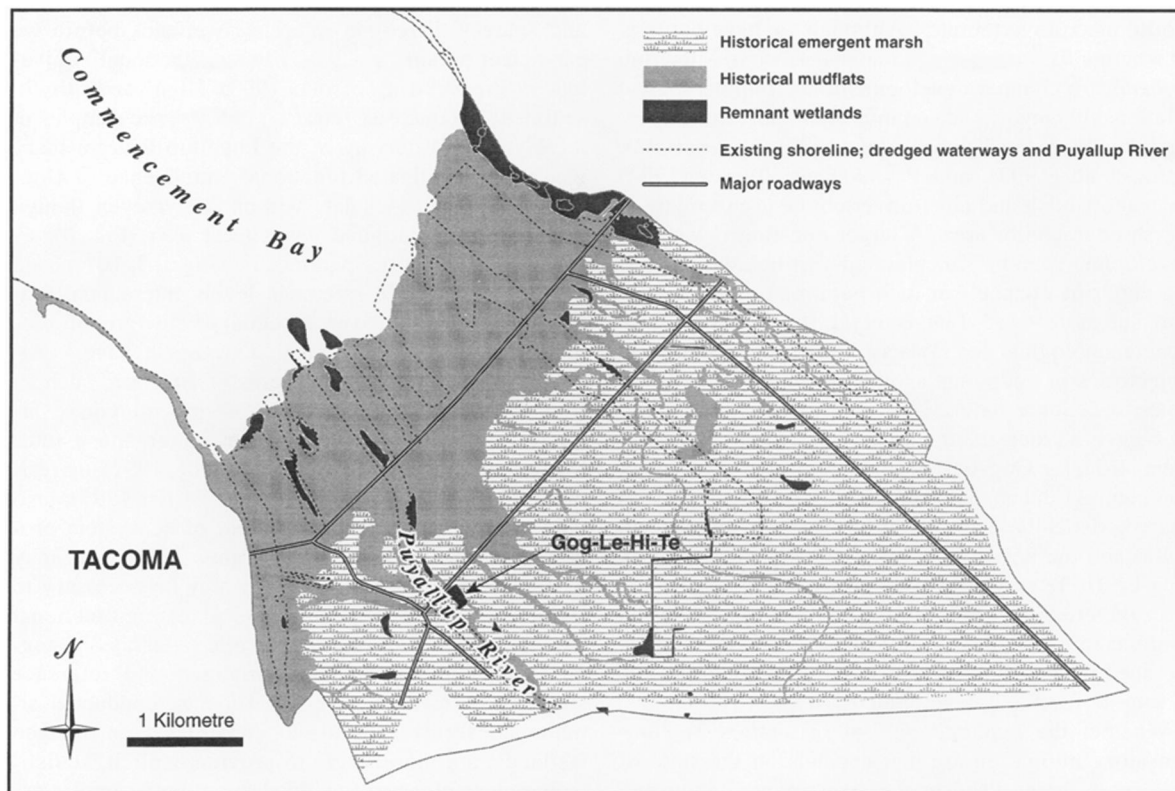


FIG. 16. The modern (ca. 1988), highly modified structure of the industrialized estuary (dashed lines) is superimposed over the pre-development (1877) structure of the estuary (gray tones). The long indentations into the former mudflat and marsh habitats are dredged slips for commercial ship access to the port facilities; the river channel, although diked and straightened from its former deltaic channel, is not dredged. The remaining small, fragmented "special aquatic sites," including wetlands, are identified as black polygons; many of these exist on shoreline fill (e.g., large polygon at middle top) of subtidal habitat, are roadside, ditchbank riparian habitats (i.e., thin elongated polygons), or reflect recent accretion (e.g., at mouth of river). Redrawn from David Evans and Associates (1991 *unpublished report* to U.S. Army Corps of Engineers Seattle District, Seattle, Washington, USA).

geochemistry. The extensive sediment accretion in Gog-Le-Hi-Te illustrates a continuing process that is likely to inhibit or delay development of a stable and mature ecosystem. As in other estuarine mitigation projects, sediments in the constructed wetland were initially coarser and compacted before accreting less consolidated silts and clays (Langis et al. 1991). Between 1987 and 1989 all channel and mid-basin habitats at Gog-Le-Hi-Te accumulated fine sediments, but three or four of the five sampling sites on intertidal flats lost fine particles (Thom et al. 1990). We perceived a net transport of fine material from the flats into the channels and basin with ebb tide and rainwater runoff from the flats. As the channels and basin filled, sedimentation probably became more homogeneous across the intertidal gradient. Sediment organic-matter content approached local reference levels on Flat 4 within 5–6 yr but remained low on Flat 5 and in the channels and the basin (Fig. 6), similar to the situation found by Moy and Levin (1991). Organic-matter deposition and accumulation in the sediment is usually correlated with vegetation type and standing stock (Lindau and Hoss-

ner 1981, Craft et al. 1988, Langis et al. 1991), which, in turn, influence infaunal and epibenthic community structure (Moy and Levin 1991). It is also evident that accretion of mineral sediments outpaced any accumulation of organic matter. Within 3 yr of the development of Gog-Le-Hi-Te, the percentage of fines was comparable at all tidal elevations sampled (only the bottom of the tidal channels appeared sandy). Thus, as in the North Carolina study, sediment structure could not be used to explain differences in the trajectories in density of infauna and epibenthic organisms.

With continued natural sediment accretion and building of intertidal flats, we tentatively predict that naturally recruited *C. lyngbyei*, *Eleocharis palustris*, and other low-elevation emergent species will advance across the flats, but the extent of their expansion will depend upon porewater salinity levels and the ultimate elevations and gradients of the flats. Similarly, development of a natural dendritic channel system is proceeding (Fig. 4), and we presume that development of a dendritic intertidal channel system will ultimately facilitate access to the intertidal marsh by fishes and

motile macroinvertebrates. Although we have not tested whether fish usage of the marsh surface is a function of dendritic channel development, such an ecological relationship may provide an important indicator of wetland development. Thomas et al. (1990), Rozas (1992), Baltz et al. (1993), and Peterson and Turner (1994) have shown fish and macroinvertebrate use to relate to marsh edge and/or area. A larger question is whether development can be "accelerated" artificially, by adding dendritic channels or incorporating organic matter into substrate (e.g., Langis et al. 1991). Short-term "ecotechnological fixes" may not produce the same trajectories or endpoints as natural, much longer-term processes. Some natural resource management agencies have advocated dredging the accumulated sediments from the Gog-Le-Hi-Te channels, despite the fact that natural tidal channels in Pacific Northwest marshes characteristically lack subtidal refugia (Simenstad 1983) and the dendritic channel pattern developing in Gog-Le-Hi-Te may ultimately promote greater fish and macroinvertebrate access to the emergent marsh. Such conflicts in wetland mitigation and restoration illustrate the need for dedicated research and ecological models of long-term marsh development.

Whether the explicit goals of restoration or compensatory mitigation are met depends on the suite of parameters being used in the assessment (Zedler and Langis 1991a and b) and the local and regional landscape structure. The premise that wetland landscape pattern determines function is virtually untested, and measures of the spatial structure of wetlands and relationship to their watersheds are seldom evaluated or required in wetland functional-assessment procedures (National Research Council 1992; D. K. Shreffler and R. M. Thom, unpublished report (1993) to Washington Department of Natural Resources, Olympia, Washington, USA; D. Smith, [U.S. Army Corps of Engineers—Waterways Experiment Station, Vicksburg, Mississippi], *personal communication*). The extremely fragmented pattern of wetlands in urbanized and industrialized landscapes logically influences the trajectories of many functional attributes. The lack of any oligohaline wetlands in the proximity of Gog-Le-Hi-Te, either up- or downriver (Fig. 16), must affect the trajectories of attributes such as natural colonization of emergent plants and non-planktonic invertebrate and fish larvae. In contrast, wetland-associated birds (Figs. 15a and b) immediately colonized the wetland, exceeding reference levels for comparable marsh habitats. Obviously, Gog-Le-Hi-Te is an oasis for wetland-associated birds in the otherwise depauperate Puyallup River delta.

Although assessing functions of restored and created wetlands should be most effective when trajectories from constructed wetlands can be compared to reference sites (Simenstad et al. 1991, Kentula et al. 1992, Richardson 1994), our results from Gog-Le-Hi-Te indicate that we need to understand natural variability

and sources thereof in reference wetlands before we can select parameters that indicate functional equivalency (Streever and Crisman 1993). High variability in sediment organic content of local reference samples in the Hylebos Waterway of the Puyallup River estuary would have indicated functional equivalence of Gog-Le-Hi-Te's intertidal flats within 1–2 yr even though accumulation continued for at least 6 yr (Fig. 6). *C. lyngbyei* production parameters (Figs. 8–10) posed even more variable reference levels interannually at one site (Chehalis River, especially belowground biomass) and among sites (e.g., Duwamish River). Although costly, explicit measures of function, such as trophic linkages to benthic and/or epibenthic communities vis a vis fish consumption patterns and rates (Moy and Levin 1991, Shreffler et al. 1992) or residence times (Shreffler et al. 1990a, Miller 1993), provide a more integrated assessment of ecosystem processes at local and landscape scales. These, too, may vary considerably. Ultimately, it may be necessary to supplant our descriptive means of assessing functional equivalency with simple, controlled "assays", standardized across restoration/mitigation and reference sites. Thus, manipulative experiments, conducted simultaneously in natural and constructed or restored wetlands, are more likely to provide explicit, statistically powerful tests of wetland function. Examples are comparisons of short-term fish foraging success or daily growth, as measured by otolith microstructure (e.g., Miller 1993) or RNA/DNA.

Our study of what is often described as one of the most "successful" estuarine restoration projects in the Pacific Northwest suggests that regulatory evaluation is precarious because of the prevailing lack of tools to predict endpoints of estuarine wetland development. It is unlikely that the currently haphazard monitoring (Kentula et al. 1992) of mitigation projects as "natural experiments" (Jordan et al. 1987) can resolve the complex, long-term patterns and processes that are required to predict outcomes. Eventually, applied studies of wetland restoration and creation need to be integrated with basic ecological studies of ecosystem succession and disturbance. Such an approach might be most effectively implemented through dedicated long-term research sites, such as the United States National Science Foundation's Long-Term Ecological Research (LTER) or Land-Margin Ecosystem Research (LMER) sites, where ecosystem processes are being examined over many scales of spatial and temporal variability.

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