Tracking Wetland Restoration: Do Mitigation Sites Follow Desired Trajectories?

Joy B. Zedler[1,2]
John C. Callaway[1]

Abstract

Hypothetical models in the scientific literature suggest that ecosystem restoration and creation sites follow a smooth path of development (called a trajectory), rapidly matching natural reference sites (the target). Multi-million-dollar mitigation agreements have been based on the expectation that damages to habitat will be compensated within 5–10 years, and monitoring periods have been set accordingly. Our San Diego Bay study site, the Sweetwater Marsh National Wildlife Refuge, has one of the longest and most detailed records of habitat development at a mitigation site: data on soil organic matter, soil nitrogen, plant growth, and plant canopies for up to 10 years from a 12-year-old site. High interannual variation and lack of directional changes indicate little chance that targets will be reached in the near future. Other papers perpetuate the trajectory model, despite data that corroborate our findings. After reviewing "trajectory models" and presenting our comprehensive data for the first time, we suggest alternative management and mitigation policies.

Introduction

The goal of ecological restoration is to return a damaged ecosystem to a more natural condition (National Research Council 1992). In setting specific objectives, restoration planners look to restoration ecologists, who in turn rely on theories of community succession and ecosystem development (Odum 1969; Grime 1977) for models of how restoration sites will change through time (Magnuson et al. 1980; Bradshaw 1984; Kentula et al. 1992; Dobson et al. 1997; cf. Fig. 1). The need to predict development rates for restored ecosystems is urgent because compensatory mitigation is becoming a widespread practice. Reducing damage to habitat is addressed in the National Environmental Policy Act (1969), the Clean Water Act (1972), and the Endangered Species Act (1973); under these policies, unavoidable damages can be compensated through habitat restoration or construction (see recent forum on wetland mitigation, Zedler 1996*a*).

Policy for compensating damages to wetlands and endangered species habitats assumes that a restored or created ecosystem will, in relatively short order, replace losses in structure and function (Environmental Protection Agency and Department of the Army 1990). Mitigation ratios—proportion of area to be restored to area damaged—are established based on how long it might take to reach the target and how closely the site is expected to match reference sites. If mitigation sites do not develop rapidly or reach their targets, a change in policy in indicated.

While many studies and simulation models characterize the temporal dynamics of vegetation composition (Bazzaz 1996), few data sets document the maturation of ecosystem processes, such as primary productivity and nutrient accumulation (Richardson 1994). Much of the predictive capability is qualitative, not quantitative (Fig. 1). Depictions of restoration site development (Magnuson et al. 1980; Bradshaw 1984; Kentula et al. 1992; Hobbs & Mooney 1993) differ from one another in several ways: (1) the frame of reference is either time (Fig. 1A, C, & D) or ecosystem structure and function (Fig. 1B, C, & D); (2) the shape of the expected pathway is straight or curved or becomes flat; (3) endpoints are single points (Fig. 1C) or include alternative outcomes (Fig. 1A, B, & D; Zimmerman et al. 1996); and (4) variation in the path is included (Fig. 1C; Richardson 1994) or not (Fig. 1A, B, & D). In all cases, however, a smooth path with increasing ecosystem function has been proposed.

In recent years, the term "trajectory" has been adopted for these hypothetical pathways (Aronson & Le Floc'h 1996; Hobbs & Norton 1996; Simenstad & Thom 1996; Dobson et al. 1997; Meffe et al. 1997). We evaluate the trajectory concept using some of the longest-term and most complete data on constructed wetland development and ask if this system follows a clear path and can be expected to hit its target. We studied a San Diego Bay mitigation site that was designed to support an endangered bird, *Rallus longirostris levipes* (the Light-footed Clapper Rail). An 8-ha area of sandy fill

[1]Pacific Estuarine Research Laboratory, San Diego State University, San Diego, CA 92182–4625, U.S.A.
[2]Current address: Aldo Leopold Professor of Restoration Ecology, Botany Department and Arboretum, University of Wisconsin–Madison, Madison, WI 53706, U.S.A.

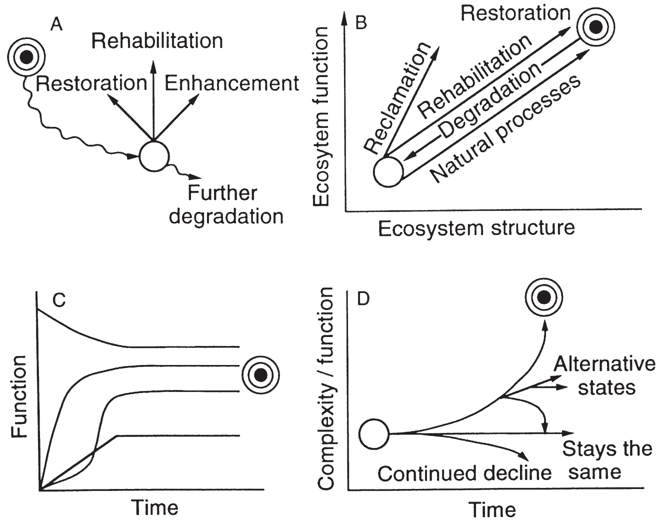

Figure 1. Hypothetical models of restoration site trajectories, with natural ecosystem conditions indicated by a bull's-eye and the degraded system as an open circle. Redrawn from Magnuson et al. (1980), this figure was developed to characterize lake degradation (A). Redrawn from Bradshaw (1984) and Dobson et al. (1997), this model characterized degradation due to mining or other operations; the author acknowledged that assistance would be needed for rapid ecosystem development (B). Redrawn from Kentula et al. (1992); the authors indicate that some attributes of constructed wetlands may initially be higher than reference systems, giving the example of Simpson's diversity index for vegetation (C). Redrawn from Hobbs and Mooney (1993) (D).

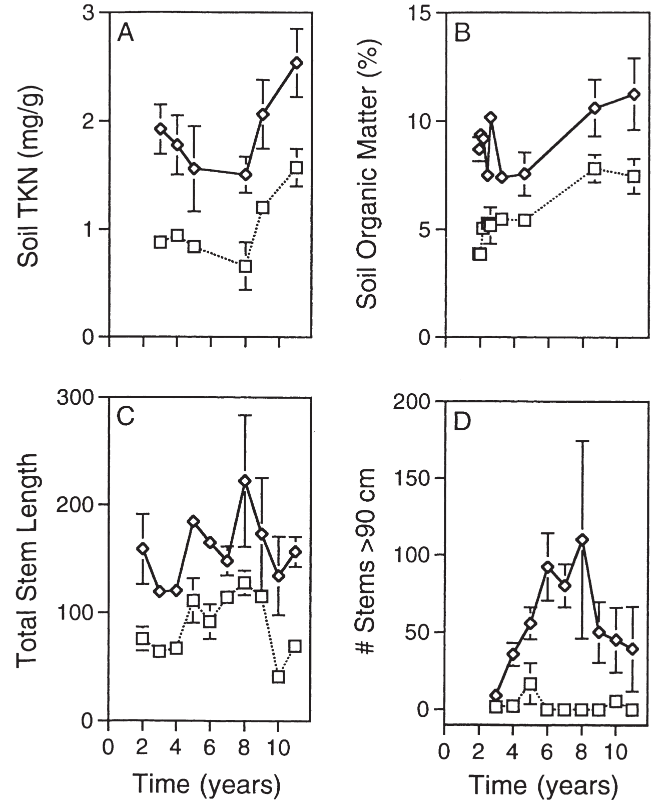

Figure 2. Changes in soil and plant canopy attributes at San Diego Bay from 2 to 11 years after planting in 1985. Squares, constructed marsh; diamonds, natural reference marsh. Total Kjeldahl nitrogen of surface soil (A). Soil organic matter (loss on ignition) (B). *Spartina foliosa* (cordgrass) total stem length (m/m^2), measured at the end of the growing season (C). Number of cordgrass stems taller than 90 cm (D).

was excavated to intertidal elevations in 1984, and eight marsh islands were planted with *Spartina foliosa* (cordgrass) in 1985 to provide nesting habitat for the clapper rail. We report data on the soil (organic matter content [OM] and total Kjeldahl nitrogen [TKN]) and the cordgrass canopy (number of tall stems and total stem length, an estimate of aboveground biomass).

Long-Term Data from San Diego Bay Marshes

In previous publications (Pacific Estuarine Research Laboratory 1990; Langis et al. 1991; Zedler 1993; Gibson et al. 1994; Boyer & Zedler 1996; Zedler 1996*b*; Haltiner et al. 1997; Boyer & Zedler 1998) we have detailed the sampling methods and results of comparing constructed and adjacent reference marshes in San Diego Bay, California. The principal target for this mitigation site is to produce tall, self-sustaining stands of cordgrass for nesting by the Light-footed Clapper Rail. The constructed wetland has a basic problem, that the substrate is much sandier than natural marshes (Langis et al. 1991). The coarse soil neither supplies nor retains sufficient nitrogen for optimal plant growth (Gibson et al. 1994; Boyer & Zedler 1998); hence, the cordgrass does not grow tall enough to support clapper rail nesting (Zedler 1993). In addition, short, stressed plants are more susceptible to insect attack than tall plants, and insect outbreaks further constrain cordgrass height (Boyer & Zedler 1996).

Organic matter content and total Kjeldahl nitrogen have proven to be useful variables for assessing soil conditions (Pacific Estuarine Research Laboratory 1990). Cordgrass growth is readily assessed as total stem length and the number of stems per square meter that are taller than 90 cm is a good indicator of suitable nesting habitat (Zedler 1993). For the first time, we report the long-term data (from years 2–11) on four soil and vegetation characteristics (Fig. 2). These data fail to support the trajectory concept for the following reasons.

First, there is a high interannual variability for both the constructed and natural marshes. This is not surprising, because southern California coastal wetlands are influenced by occasional, unpredictable floods that deliver fresh water, nutrients, and sediment in pulses. Both biomass and cordgrass height are known to respond positively to flooding (Zedler 1983), which occurred twice during the monitoring period, in 1993 and 1995 (years 8 & 10). High interannual variability was also reported for a constructed wetland near Puget Sound, where most of the assessed attributes failed to show directional change (Simenstad & Thom 1996).

Second, none of our data indicate strong directional trends in ecosystem development, although the data for soil OM suggest a weak trajectory. Because all attributes measured in our reference marsh showed interannual variability, we relativized the data by expressing values in the constructed marsh as a proportion of those in the natural marsh (Fig. 3). Curves fit to the relativized data did not match hypothetical paths (Fig. 1), and the fit of linear equations to these data (suggested by Fig. 1B) was poor ($r^2 < 0.35$ for all four linear regressions). Soil OM increased early and quickly reached a plateau. If we assume a linear relationship for the asymptotic increase in soil OM, we predict that it will be 22 years before the sites are equivalent; an exponential equation fits the data better than the linear model, however, with an asymptote at 73% of natural marsh conditions. The curve for sediment TKN inclines slightly, but extrapolation indicates that it will take more than 40 years to intercept levels in reference marshes.

Third, even where an incline is indicated, the target will not be intercepted in the short term. The sandy,

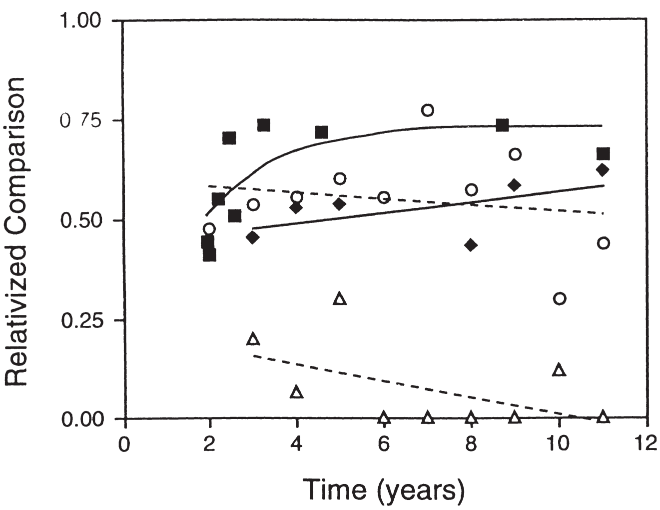

Figure 3. Relativized (constructed/natural) values for attributes in Figure 2. Regressions were as follows: total Kjeldahl nitrogen of soil (filled diamonds): $y = 0.438 + 0.013 *$ years ($r^2 = 0.35$); soil organic matter (filled squares): $y = 0.73 * (1 - e^{-0.61 * \text{years}})$ ($r^2 = 0.98$; corrected $r^2 = 0.59$), linear regression line, $y = 0.520 + 0.021 *$ years ($r^2 = 0.27$), not shown; cordgrass total stem length (open circles): $y = 0.599 - 0.008 *$ years ($r^2 = 0.035$); cordgrass stems >90 cm (open triangles): $y = 0.219 - 0.021 *$ years ($r^2 = 0.26$); where y = the relativized score for each parameter.

dredge-spoil sediment fails to supply or retain nitrogen (Langis et al. 1991). Our findings of slow nutrient accumulation corroborate spatial patterns in nitrogen accumulation among marshes of different age in North Carolina (Craft et al. 1988).

Finally, although we can calculate an expected time for equivalency with reference sites, this is unwise for ecosystems that are highly responsive to pulsed events (Odum et al. 1995). Year-to-year differences can occur in either the rate of change (faster or slower) or its direction (toward or away from the target): for example, 10 cm of accumulated sediments can shift tidal marsh composition to a different assemblage of plant species. Where there is high interannual variability, the time to functional replacement should not be extrapolated from short-term observations.

Discussion

We conclude from our long-term study that the constructed marsh has not met agency expectations for compliance with mitigation criteria for the endangered Light-footed Clapper Rail within the short term (i.e., the usual 5–10 monitoring framework of regulatory agencies), and we predict that the constructed marsh soil will not match that of natural wetlands in the long term (regression analysis predicted equivalency in TKN levels after more than 40 years; OM leveled off at 75% of the target). A Pennsylvania study of 44 wetland restoration projects aged 1–8 years also failed to identify progressive change in soil organic matter (Bishel-Machung et al. 1996). Minello and Webb (1997) assessed 10 created salt marshes ranging from 3 to 15 years old; sediment macro-organic matter increased over time, but age explained only 9% of the variability. In our research, two vegetation attributes, total stem length and the density of tall stems, declined in relation to the reference site, and we predict that the canopy will not become suitable for nesting by the endangered Light-footed Clapper Rail. This bird has yet to nest in marshes constructed from dredge spoil.

We disagree with Aronson and Le Floc'h (1996:385) that "any and all changes an ecosystem undergoes in a given period of time can be seen in comprising a trajectory" and that "trajectory seems a useful term for describing what happens to an ecosystem undergoing ecological restoration or rehabilitation." The assumption that much will be achieved rapidly is not appropriate for highly modified sites. To date, time frames for ecosystem replacement have been based on spatial comparisons (e.g., constructed wetlands of differing age) from only a few habitat types (e.g., systems dominated by nonwoody vegetation). Reconstruction times for old-growth forests and their soil properties are unknown. Even for herbaceous wetlands, we would expect functional replacement within 5–10 years only for low-stress systems (e.g., cattail marshes with substantial nutrient input). Species-rich systems, or those with special water-quality requirements (e.g., fens) are likely to take much longer, if functional equivalency can be achieved. With better predictions of development times (20–100 years), regulatory agencies could specify more realistic standards for ecosystem replacement, and mitigators could better plan the nature and length of their mitigation projects. Because the time scales for ecosystem development often extend beyond those acceptable to mitigators, regulators should always strive to prevent damages to critical ecosystems rather than to permit losses and hope for compensation.

Because mitigation sites are often located in urban areas, several factors—initial conditions, modified hydrology, exotic species invasions, effects of feral animals—are likely to interfere with ecosystem- and community-development trends. Resource managers who must set standards for mitigating damages to wetlands and endangered species need a new paradigm for sites in highly stressed environments and guidelines for deciding what to do if criteria are unlikely to be met.

We suggest that mitigation policy include recognition that (1) compensation sites may never fully replace natural wetland functions, (2) the time to functional equivalency may well exceed the usual monitoring periods, and (3) long-term predictions of the time to functional equivalency may not be meaningful if they are based on short-term data from pulse-driven ecosystems. Where functional replacement is unlikely, either the permit for the proposed damages should be denied or mitigation ratios should be adjusted in relation to the maximum relativized value (Fig. 3) that a mitigation site is expected to attain in the short term. At present, compensation ratios range from below 1:1 to replace highly degraded habitat to as high as 10:1 to compensate damages to mature wetland replaced by creating wetland of unknown potential. Higher ratios should become standard for replacing ecosystems that require longer development periods or that have not been replaced in previous restoration efforts. While the general goal of matching the structure and function of the impact site or a suitable reference wetland is easy to state, the reality is that ecosystem development may proceed along complex paths that are difficult or impossible to predict, given the shortcomings of restoration sites.

Some restoration sites or ecosystem-development attributes may follow trajectories. Smooth and rapid change may be expected for restoration projects in landscapes that are more intact and where damages are less severe (National Research Council 1992). The restoration of a river that has been temporarily diverted may progress steadily as seed banks regenerate and animals gradually return (Cummins & Dahm 1995). Mitigation projects in the most degraded sites (e.g., urban habitats) with the most difficult targets (e.g., support for endangered species) and with a high degree of environmental pulsing are less likely to proceed smoothly toward replacement of the habitat values of natural wetlands. To assess the generality of our conclusion, we recommend studies of ecosystem development in a variety of less-degraded landscapes.

LITERATURE CITED

Aronson, J., and E. Le Floc'h. 1996. Vital landscape attributes: missing tools for restoration ecology. Restoration Ecology **4:** 377–387.

Bazzaz, F. A. 1996. Plants in changing environments: linking physiological, population, and community ecology. Cambridge University Press, New York.

Bishel-Machung, L., R. P. Brooks, S. S. Yates, and K. L. Hoover, 1996. Soil properties of reference wetlands and wetland creation projects in Pennsylvania. Wetlands **16:**532–541.

Boyer K. E., and J. B. Zedler. 1996. Damage to cordgrass by scale insects in a constructed salt marsh: effects of nitrogen additions. Estuaries **19:**1–12.

Boyer, K. E., and J. B. Zedler. 1998. Effects of nitrogen additions on the vertical structure of a constructed cordgrass marsh. Ecological Applications **8:**692–705.

Bradshaw, A. D. 1984. Ecological principles and land reclamation practice. Landscape Planning **11:**35–48.

Craft, C. B., S. W. Broome, and E. D. Seneca. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. Estuaries **11:**272–280.

Cummins, K. W., and C. N. Dahm. 1995. Introduction: restoring the Kissimmee. Restoration Ecology **3:**147–148.

Dobson, A., A. D. Bradshaw, and A. J. M. Baker. 1997. Hopes for the future: restoration ecology and conservation biology. Science **277:**515–522.

Environmental Protection Agency and Department of the Army. 1990. Memorandum of Agreement between the Environmental Protection Agency and the Department of the Army Concerning the Determination of Mitigation under the Clean Water Act Section 404(b)(1) Guidelines. Signed by Robert W. Page (Assistant Secretary of the Army) and LaJuana S. Wilcher (Assistant Administrator of Water, EPA) on February 6, 1990, Washington, D.C.

Gibson, K. D., J. B. Zedler, and R. Langis. 1994. Limited response of cordgrass (*Spartina foliosa*) to soil amendments in constructed salt marshes. Ecological Applications **4:**757–767.

Grime, J. P. 1977. Evidence for the existence of three primary strategies in plants and its relevance to ecological and evolutionary theory. American Naturalist **982:**1169–1194.

Haltiner, J., J. B. Zedler, K. E. Boyer, G. D. Williams, and J. C. Callaway. 1997. Influence of physical processes on the design, functioning and evolution of restored tidal wetlands in California. Wetlands Ecology and Management **4:**73–91.

Hobbs, R. J., and H. A. Mooney. 1993. Restoration ecology and invasions. Pages 127–133 in D. A. Saunders, R. J. Hobbs, and P. R. Erlich, editors. Nature Conservation 3: reconstruction of fragmented ecosystems, global and regional perspectives. Surrey Beatty and Sons, Chipping Norton, New South Wales, Australia.

Hobbs, R. J., and D. A. Norton. 1996. Towards a conceptual framework for restoration ecology. Restoration Ecology **4:** 93–110.

Kentula, M. E., R. P. Brooks, S. E. Gwin, C. C. Holland, A. D. Sherman, and J. C. Sifneos. 1992. An approach to improving decision making in wetland restoration and creation. U.S. Environmental Protection Agency, Corvallis, Oregon.

Langis, R., M. Zalejko, and J. B. Zedler. 1991. Nitrogen assessment in a constructed and a natural salt marsh of San Diego Bay. Ecological Applications **1:**40–51.

Magnuson, J. J., H. A. Regier, W. J. Christie, and W. C. Sonzogni. 1980. To rehabilitate and restore Great Lakes ecosystems. Pages 95–112 in J. Cairns, Jr., editor. The recovery process in damaged ecosystems. Ann Arbor Science, Ann Arbor, Michigan.

Meffe, G. K., C. R. Carroll, and Contributors. 1997. Principles of conservation biology. 2nd edition. Sinauer Associates, Inc., Sunderland, Massachusetts.

Minello, T. J., and J. W. Webb, Jr. 1997. Use of natural and created *Spartina alterniflora* salt marshes by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series **151:**165–179.

National Research Council. 1992. Restoration of aquatic ecosystems. National Academy of Science, Washington, D.C.

Odum, E. P. 1969. The strategy of ecosystem development. Science **164:**262–270.

Odum, W. E., E. P. Odum, and H. T. Odum. 1995. Nature's pulsing paradigm. Estuaries **18:**547–555.

Pacific Estuarine Research Laboratory. 1990. A manual for assessing restored and natural coastal wetlands with examples from southern California. Report no. T-CSGCP-021. California Sea Grant College, La Jolla, California.

Richardson, C. J. 1994. Ecological functions and human values in wetlands: a framework for assessing forestry impacts. Wetlands **14:**1–9.

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

Zedler, J. B. 1983. Freshwater impacts in normally hypersaline marshes. Estuaries **6:**346–355.

Zedler, J. B. 1993. Canopy architecture of natural and planted cordgrass marshes: selecting habitat evaluation criteria. Ecological Applications **3:**123–138.

Zedler, J. B. 1996*a*. Ecological issues in wetland mitigation: an introduction to the forum. Ecological Applications **6:**33–37.

Zedler, J. B. 1996*b*. Tidal wetland restoration: a scientific perspective and southern California focus. Report T-038. California Sea Grant College System, University of California, La Jolla, California.

Zimmerman, J. K., M. R. Willig, L. R. Walker, and W. L. Silver. 1996. Introduction: disturbance and Caribbean ecosystems. Biotropica **28:**414–423.