Environmental Science                                          College of Arts and Sciences

2005

# The Challenge of Restoring Functioning Salt Marsh Ecosystem

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**ABSTRACT**

CALLAWAY, J.C., 2005. The challenge of restoring functioning salt marsh ecosystems. *Journal of Coastal Research*, SI(40), 24–36. West Palm Beach (Florida), ISSN 0749-0208.

Substantial improvements have been made in the restoration of coastal salt marshes over the last decade; however, many challenges remain. Some opportunities for improving restoration efforts include: I. Increasing our understanding of the development of restored salt marsh ecosystems over time, especially in comparison to natural marsh development; and identifying the limiting factors that restrict the development of restored salt marshes. II. Considering the role of plant species diversity in restored salt marshes. Recent research at Tijuana Estuary has demonstrated that there is a significant effect of plant species diversity on the development of ecosystem functions in a restored salt marsh; further study of these effects is warranted in other salt marsh ecosystems. III. Evaluating the link between physical heterogeneity and ecosystem function. Small-scale changes in physical factors, such as elevation or hydrology, are likely to have substantial effects on the development of ecosystem function in restored salt marshes, and these factors should be considered in restoration design. IV. Addressing the potential impacts of exotic plants within restored marshes. Exotic species remain a substantial problem in many restored ecosystems; better efforts are needed to identify appropriate methods to control exotic plants. V. Incorporating scientific approaches into restoration efforts. Rigorously designed scientific experiments that identify cause-effect relationships for the development of restored salt marshes could substantially improve the design, implementation, and monitoring of restoration projects.

ADDITIONAL INDEX WORDS: *Ecosystem functions, trajectories, species diversity, heterogeneity, exotic species, wetland restoration.*

## INTRODUCTION

There has been growing interest in restoring wetlands in the United States over the last few decades, both as a result of mitigation related to the Clean Water Act (NATIONAL RESEARCH COUNCIL, 2001) and due to efforts targeted to increase habitat that are funded by either private organizations or government agencies. Coastal salt marshes have received substantial attention for restoration largely because of their close proximity to large population centers, as well as their importance to coastal fisheries and other ecosystem functions. With this interest, there have been a number of reviews over the last decade that have evaluated progress and made recommendations for improvement. For example, multiple reviews of coastal salt marsh restoration were included in the groundbreaking book by KUSLER and KENTULA (1990). Special journal issues on coastal restora-

tion have focused on mitigation (ZEDLER, 1996a), goal setting and success criteria (CAIRNS, 2000; HACKNEY, 2000), physical and ecological processes of tidal wetland restoration (GOODWIN *et al.*, 2001), and dike/levee breaching for coastal marsh restoration (SIMENSTAD and WARREN, 2002).

Across these reviews there have been some consistent recommendations, including the improvement of pre-project planning and a focus on clear goals, a consideration of regional issues, and an improvement of monitoring efforts. Advancements have been made in each of these areas that have substantially improved restoration efforts. In terms of pre-project planning, one of the critical issues has been the push for clearer goals that can be addressed by measurable attributes. Another planning concern has been to establish goals based on ecosystem functions rather than structure. While we will probably continue to monitor primarily structure, given that it is easier to measure, efforts have been made to link structural attributes to ecosystem functions. Finally there has

been an attempt to make a strong connection between project goals, parameters of interest, and monitoring methods so that projects can be evaluated based on *a priori* criteria.

Substantial improvement has been made in the adoption of regional goals and approaches. For example, ZEDLER (1996b) emphasized the need for this issue in Southern California, and since then there has been the creation of a regional collaboration between federal, state, and local agencies to improve regional planning, called the Southern California Wetlands Recovery Project (SCWRP; see www.scc.ca.gov). Many regional restoration projects have been funded in the last five years by SCWRP, and there has been ongoing scientific input to SCWRP. Efforts also have been undertaken to develop a regional approach in San Francisco Bay, where the San Francisco Habitat Goals Project (GOALS PROJECT, 1999) brought together a broad range of scientists to identify the needs of salt marsh species from plants to invertebrates, fish, birds, mammals, reptiles, and amphibians. The project was designed to set long-term regional plans for restoration so that it would not be driven by a case-by-case consideration of mitigation projects.

In other areas, regional or large-scale approaches have also been established. In Louisiana, the Coastal Wetlands Planning, Protection, and Restoration Act (CWPPRA) has funded many projects (STEYER and LLEWELLYN, 2000), and this has led to the development of additional efforts to coordinate restoration across the region (*e.g.,* see www.lacoast.gov). Restoration efforts in the Everglades also have focused on a coordinated, large-scale approach (www.evergladesplan.org). In Delaware, one of the single largest coastal restoration projects in the country has received substantial scientific and planning interest (WEINSTEIN *et al.,* 2001).

Finally, there have been improvements in monitoring efforts. Whereas most early restoration efforts were monitored for only one to three years, recent mitigation projects are commonly monitored for three to five years according to literature and testimony provided to the National Research Council Committee that reviewed compensatory mitigation issues (NATIONAL RESEARCH COUNCIL, 2001). In addition, there has been more incorporation of adaptive management into many restoration efforts (STEYER and LLEWELLYN, 2000; THOM, 2000).

## CURRENT CHALLENGES

Despite the improvements outlined above, many challenges remain for salt marsh restoration. Even with the best intentions, a number of projects fail to provide functioning, sustainable ecosystems. In order to continue moving the field of restoration ecology forward, scientists need to identify and address the major constraints on restoration implementation and policy. Significant areas that offer opportunities for improving restoration efforts include:

I.  Increasing our understanding of the development of restored salt marsh ecosystems over time,
II. Considering the role of species diversity in restored salt marshes,
III. Evaluating the link between physical heterogeneity and ecosystem functions,
IV. Addressing the potential impacts of exotic species within restored marshes, and
V.  Incorporating experimentation into restoration.

Each of these issues is discussed in more detail below.

### I. The Development of Restored Salt Marsh Ecosystems Over Time

One of the overarching concerns of wetland restoration research is to understand what controls the development of ecosystem functions over time. A substantial effort has been made to understand the conceptual development of ecosystem functions within the framework of succession theory, resulting in the use of "trajectories" for analyzing this development. A range of approaches using hypothetical trajectories of ecosystem development have been proposed for restored ecosystems (MAGNUSON *et al.,* 1980; BRADSHAW, 1984; KENTULA *et al.,* 1992; HOBBS and MOONEY, 1993; DOBSON *et al.,* 1997). KENTULA *et al.* (1992) formulated possible trajectories for restored wetlands, and this approach has been adopted in evaluating the development of restored salt marshes. However, testing the trajectory concept for restoration development has been a challenge because it requires long-term data for both restored and natural reference marshes, something that is rarely available, given the short history of restoration and the lack of monitoring for many early projects. It is only in the last few years that trajectories have been evaluated for more than a small number of sites (Table

Table 1. *Compilation of salt marsh restoration projects that have used the trajectory approach in evaluating the development of ecosystem attributes. OM = organic matter, N = nitrogen.*

| Location | Num-ber of Sites | Parameters Measured | | | | | | Monitoring Period | Reference |
|---|---|---|---|---|---|---|---|---|---|
| | | Soil OM | Soil N | Plants | Inverts | Fish | Birds | | |
| Gog-Le-Hi-Te, Wetland, WA | 1 | × | | × | × | × | × | 1–7 years | SIMENSTAD and THOM, 1996 |
| Pine Knoll and Snow's Cut, NC | 2 | × | × | × | × | | | 1–25 years | CRAFT et al., 1999 |
| San Diego Bay, CA | 1 | × | × | × | | | | 2–11 years | ZEDLER and CALLAWAY, 1999 |
| Salmon River Estuary, OR | 3 | | | | × | × | | 3–21 years | GRAY et al., 2002 |
| Sarah's Creek, VA | 1 | × | | × | × | × | × | 5 and 12 years | HAVENS et al., 2002 |
| Great Bay Estuary, NH | 6 | × | | × | | | | 1–14 years | MORGAN and SHORT, 2002 |
| Long Island Sound, CT | 9 | | | × | × | × | × | 1–21 years | WARREN et al., 2002 |

1). Trajectories have been used to evaluate a range of ecosystem attributes from soil properties to plants and animals (Table 1).

In evaluating the trajectories from these various studies, it's clear that there is a wide range of results from this approach. SIMENSTAD and THOM (1996) were the first to use this approach to assess a restored estuarine wetland, the Gog-Le-Hi-Te Wetland in the Puget Sound. They measured multiple ecosystem attributes over a seven-year period (including soil, sediment, productivity, invertebrates, fish, and birds) with mixed results across the various attributes. Total invertebrate species richness increased as well as the diversity of associated fishes during this time period; however, these were the only two of sixteen parameters that followed such trajectories (SIMENSTAD and THOM, 1996). In a longer-term assessment, CRAFT et al. (1999) used a 25-year record of a restored salt marsh in North Carolina and found that aboveground biomass and macro-organic matter in the restored marsh reached equivalency with the reference site within 10 years, while the benthic invertebrate community took 15–25 years to reach this level. Soil carbon and nitrogen reserves were still well below the natural marsh levels after 25 years (CRAFT et al., 1999). ZEDLER and CALLAWAY (1999) evaluated plant and soil characteristics at a created salt marsh in San Diego Bay, California, and found little support for the development of trajectories based on an 11-year data set. Created marshes and highly degraded sites represent the greatest challenges for restoration. Only soil nitrogen concentrations showed a continual increase at the created wetland (relative to a reference site), and at the measured rate of increase it would take over 40 years for the mitigation site to equal conditions at the nearby reference wetland. ZEDLER and CALLAWAY (1999) used a "relativized" index of

the attributes (the restored wetland value divided by the reference wetland value) to compensate for annual variation in conditions. Plant biomass developed relatively quickly within a created brackish-water marsh in North Carolina, but soil organic carbon could take 100 to 200 years to develop levels similar to nearby natural marshes (CRAFT et al., 2002). The development of carbon and other soil characteristics within the created brackish marsh was strongly affected by the duration of inundation, with the slowest rates of development occurring at high elevations (CRAFT et al., 2002).

A common approach that has been used to side-step the challenge of long-term data has been to use a "space-for-time" substitution: simultaneously evaluating conditions at a number of sites of different ages, rather than considering the development of a single site over time (PICKETT, 1991). GRAY et al. (2002), MORGAN and SHORT (2002), and WARREN et al. (2002) have all used this approach, with support for the trajectory approach. MORGAN and SHORT (2002) point out that the use of multiple sites may add variability to the analyses; however, they felt that the space-for-time substitution was useful. MORGAN and SHORT (2002) discuss the important difference between the application of trajectories for created wetlands versus restored wetlands. POACH and FAULKNER (1998) evaluated phosphorus dynamics in created dredge-material wetlands and used a slightly different approach, comparing newly restored sites to newly developing natural sites. TYLER and ZIEMAN (1999) evaluated the development of a natural salt marsh using a trajectory approach and also summarized a number of trajectory evaluations of both natural and restored wetlands.

While trajectories are very useful as a general framework in evaluating the development of restored salt marshes and other ecosystems, there

are still challenges to consider in applying trajectories to restoration and management decisions. Generalizations about overall ecosystem development based on a small number of attributes are not possible. Different parameters are likely to follow different trajectories, and these may not be consistent from one marsh to another. Soil conditions are likely to be the slowest parameters to develop at restored or created salt marshes (CRAFT et al., 1999; ZEDLER and CALLAWAY, 1999). Furthermore, trajectories will be affected by initial site conditions and may not always be highly predictable. Some particular issues that need further consideration in the interpretation and application of trajectories include the following.

(1) Consider how degraded sites develop over time. ZEDLER (1999) hypothesized that highly degraded sites are likely to be less predictable in their development and will take a longer time to develop comparable levels of ecosystem function. Further, some sites may be so degraded that they have gone through a threshold change and will never reach the level of functioning that is found for similar natural wetlands or will develop into an alternative state (DOBSON et al., 1997). Similarly, sites that have been degraded for extended periods are likely to be more difficult to restore and to have a higher degree of variability in their development. Additional information is needed from salt marshes that cover the spectrums of degradation and time scales to gain better insight into how these factors may affect restoration possibilities and the development of ecosystem functions over time.

(2) Collect additional data to evaluate trajectories. As noted above, the longest data set that currently exists for salt marsh restoration projects is 25 years (CRAFT et al., 1999). Of course, we will continue to grow longer-term data sets, but this is a very slow process. More evaluations of restoration development using "space-for-time" substitutions (PICKETT, 1991) should be completed. Although scientific restoration has only taken place for the last 20–30 years, there are some examples of both intentional and unintentional restoration over much longer time-scales, in particular levee breaches (e.g., some levee failures in San Francisco Bay date back 70 years or more and similar examples can be found in Louisiana and other areas). These long-term "restoration" projects offer a unique opportunity to evaluate sites over a much broader time scale. If these sites are selected carefully and combined with existing datasets from

restoration projects that have been monitored continuously, this would give insight into the longer-term development of restored salt marshes.

In addition to long-term data sets, more information is needed from recently restored sites to evaluate their short-term, immediate development. Many of the interesting differences in trajectories are likely to occur in the early stages of development; however, without data from a number of different sites it will be difficult to improve our understanding of this phase of development.

(3) Use "trajectories" to identify limiting factors for salt marsh development. The combination of trajectories with more detailed evaluations of sites to identify the limiting factors for development is critical. We need to know much more than just whether a site is likely to develop over time. Instead, the question that we really should focus on is: what is restricting the development of function at a site? To address this question, we need to combine the trajectory approach with experimental evaluations of restored sites (see V. Incorporation of Experimentation into Restoration) since it is only with manipulative experiments that we can clearly address cause and effect relationships in the development of restored ecosystems.

(4) Consider the policy implications of trajectories. Although the trajectory research completed to date has focused primarily on the issue from a scientific perspective, there is also substantial interest in applying this information to improve policy decisions. For example, as more data are evaluated, trajectory analyses may help to identify the time period that is necessary for the development for various ecosystem attributes, and this information would be very useful in establishing the appropriate time period for monitoring of salt marsh mitigation projects. Trajectory analysis also may help to identify ecosystem attributes that are most important to consider in the early stages of ecosystem development versus those that need longer-term study. For attributes that have a highly predicable trajectory, it may be possible to reduce the period or frequency of compliance monitoring if the attribute is following an acceptable pathway. Given these policy interests, we need to consider trajectory variability and how reliable predictions might be based on early monitoring and trajectory analysis.

## II. The Role of Species Diversity in Restored Salt Marshes

Evaluating the link between species diversity and ecosystem function has seen an explosion of
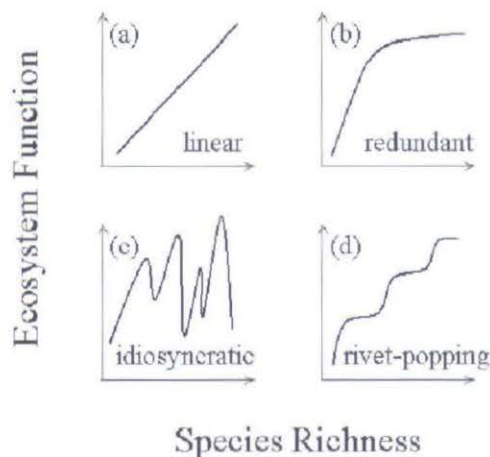
Species Richness

Figure 1. Proposed relationships between species richness and ecosystem function (modified from Naeem, 1998). Linear (a) indicates a relationship between species richness and function in which each species counts equally. Redundant (b) and rivet-popping (d) indicate that loss or addition of particular species or functional groups cause critical changes in ecosystem function. Idiosyncratic (c) indicates no relationship between diversity and ecosystem function.

interest in the last decade (*e.g.*, SCHULZE and MOONEY, 1993; NAEEM *et al.*, 1994; TILMAN *et al.*, 1997; LOREAU *et al.*, 2001; TILMAN *et al.*, 2001). This research has focused on answering a series of questions concerning the potential relationships between species diversity and function (Figure 1), including: (1) does species richness (the number of species present) affect ecosystem function, and (2) does species composition (the identity of species present) affect ecosystem function? Debate over these issues has been substantial, with much concern over the design of experiments and data interpretation (HUSTON, 1997; TILMAN *et al.*, 1997; LOREAU *et al.*, 2001; NAEEM, 2002). The debate is of particular interest because there are important implications for the management and restoration of ecosystems. For example, the nature of the relationship between diversity and ecosystem function may direct our efforts toward preserving or restoring all species within an ecosystem (as indicated by the linear relationship where all species have equal importance) or toward identifying particular critical species within that ecosystem (as indicated by the redundant or rivet-popping relationship where composition or functional groups are most important) (Figure 1).

Most of the diversity-function research has fo-

cused on grassland ecosystems that are species rich (TILMAN *et al.*, 1997; HECTOR *et al.*, 1999). While there is interest in evaluating the relationship between function and diversity in these species-rich ecosystems, there also are some challenges due to high levels of richness. For example, if species combinations are chosen intentionally for the treatments, there will be limitations in data interpretation since effects may be due to individual species and not to the number of species that are present (HUSTON, 1997). Furthermore, if the species pool is large, and species are chosen randomly, it is unlikely that the combinations of species that are chosen will represent "real" assemblages that are actually found in the field. Salt marsh research can add to the diversity-function debate because only a small number of plants are in the species pool, and randomly chosen combinations for an experiment are much more likely to actually occur in the field (ZEDLER *et al.*, 2001; CALLAWAY *et al.*, 2003). Little research has been completed in other ecosystems with a small number of species (*e.g.*, ENGELHARDT and RITCHIE, 2001), despite the call for diversity experiments that focus on one to ten species (VITOUSEK and HOOPER, 1993).

Furthermore, almost all of the diversity research that has been completed to date comes from a conservation perspective, asking the question: what happens to an ecosystem if we lose species due to extinction or local extirpation? From a restoration perspective, the issue of diversity is also of substantial interest; however, the key question is slightly different: how many species do we need to include in a restoration project to achieve a particular level of ecosystem function (ZEDLER *et al.*, 2001; CALLAWAY *et al.*, 2003). This question is of substantial concern because many restored ecosystems lack species diversity. The key issue for many restoration projects has simply been to establish plant cover, and the focus has been on easy-to-establish species. Species that may be difficult to establish are often not included in basic planting designs, unless they are targeted for a particular reason (*e.g.*, an endangered status or an important plant for an animal species of interest). For example, in many restored salt marshes in California, the focus for planting in the salt marsh plain has been on *Salicornia virginica* (pickleweed). While this species is dominant, there are other species that also are important for various ecosystem functions within the salt marsh plain (SULLIVAN and ZEDLER, 1999). Furthermore, when this

species is planted alone it can outcompete other species, leading to restoration sites with low plant diversity. Given these types of concerns, there is a real need to thoroughly evaluate the link between diversity and function in restored ecosystems.

Recent research at the Tidal Linkage at the Tijuana Estuary has demonstrated that there is a significant effect of plant species diversity on the development of ecosystem functions in a restored salt marsh. Experimental plots were established with 0, 1, 3, and 6 species that were randomly chosen from the pool of the eight most common plants that occur on the marsh plain in Southern California. The plots with 6 species accumulated more biomass and nitrogen than 0- and 1-species assemblages, with 3-species assemblages being intermediate, indicating that species richness has an effect on the development of marsh functions (CALLAWAY *et al.*, 2003). Individual species also affected dynamics, with the local dominant, *S. virgnica*, contributing the most biomass in plots where it was planted, while *Triglochin concinna* had the highest tissue nitrogen concentrations. Overall plant cover was similar across plots, but assemblages with multiple species developed canopies that were more complex, *i.e.*, these canopies had more layers (KEER and ZEDLER, 2002). Based on these results, manipulating species richness at the time of restoration planting can be an effective tool for accelerating the rate of functional development of salt marshes (ZEDLER *et al.*, 2001; CALLAWAY *et al.*, 2003).

## III. The Link Between Physical Heterogeneity and Ecosystem Functions

Simply having a diversity of plant species or other marsh components is not enough to ensure a high level of ecosystem function; the spatial arrangement of these components within the marsh also affects the development of ecosystem function. Physical heterogeneity, primarily in elevation and hydrology, is likely to be important across a variety of scales and will drive heterogeneity in biological processes by creating a range of physical conditions for plants and animals (*e.g.*, period of inundation, degree of soil drainage, soil salinity, *etc.*). Evaluation of the relationships between physical and biological heterogeneity has focused on plants, since plants are stationary and create the physical structure of the habitat; this is important for animal use of restored marshes. In particular, topographic heterogeneity has been shown

to affect plant distributions and overall plant diversity in a variety of wetland ecosystems (BERTNESS and ELLISON, 1987; VIVIAN-SMITH, 1997; ZHANG *et al.*, 1997; ZEDLER *et al.*, 1999; SANDERSON *et al.*, 2000).

Within salt marshes, tidal creeks are the primary source of heterogeneity and can vary in width and depth over a wide range of scales. In many cases, natural levees develop adjacent to tidal creeks, with slightly higher elevations, coarser sediments, and better drainage. Smaller creeks may lack natural levees but still create slightly different physical conditions, with particular effects on soil drainage. SANDERSON *et al.* (2000) showed that even small creeks (only 50 cm wide) have significant effects on vegetation distributions, with an average of 1.6 more plant species found in areas within 10 m of creeks. ZEDLER *et al.* (1999) evaluated the effects of creeks on plant distribution in a salt marsh in San Quintín Bay, Baja California, Mexico, and found that four plant species occurred at lower elevations when they were growing adjacent to tidal creeks, resulting in greater species richness in areas near creeks. It was hypothesized that this difference was due to better drainage adjacent to creeks. Shifts in plant species distribution also could be due to other processes, including historical impacts such as storm disturbances (GRACE and GUNTENSPERGEN, 1999), seed banks (HOPKINS and PARKER, 1984), and a variety of biological interactions (BERTNESS and ELLISON, 1987; HACKER and BERTNESS, 1999), all of which are likely to vary with spatial shifts in physical factors.

In addition to affecting the distributions of individual plant species, heterogeneity associated with creeks is likely to affect primary productivity, habitat structure, and other plant characteristics (NIERING and WARREN, 1980; WIEGERT *et al.*, 1983). For example, on the East and Gulf coast, drainage is one of the key factors controlling growth forms of *Spartina alterniflora* (MENDELSSOHN, 1979; BURESH *et al.*, 1980), with much greater productivity in tall forms that are found growing adjacent to tidal creeks (WIEGERT *et al.*, 1983; MITSCH and GOSSELINK, 2000). Tidal creeks also affect organisms besides plants. Shrimp and blue crab were most abundant within 1 m of the marsh-creek edge and declined rapidly away from the edge of the marsh in a natural marsh in Texas (MINELLO and ROZAS, 2002). Many fish and crustacean species are associated with this marsh-water interface or "marsh edge," and marshes with a greater amount of marsh edge are likely to support higher levels

of benthic infauna, as well as fish productivity (MI-
NELLO and ROZAS, 2002; WHALEY and MINELLO,
2002).

Topographic features such as pannes, hum-
mocks, mounds, and berms also create shifts in el-
evation that can have substantial effects on hy-
drology, soil chemistry, and other physical condi-
tions. These features may range in size from
small-scale hummocks that are associated with
particular plant species (e.g., Spartina patens (NY-
MAN et al., 1995)) to large pannes (NIERING and
WARREN, 1980; BERTNESS and PENNINGS, 2000).
While creeks have been well studied for their im-
pacts on physical factors, little has been done to
address the physical effects of these features.

Given that heterogeneity is important in natural
marshes across a range of scales and processes, it
should be carefully considered in restoration de-
sign and implementation. However, little research
has been done within a restoration framework to
evaluate the importance of physical heterogeneity.
A large-scale experiment concerning these impacts
is currently underway at the Model Marsh in the
Tijuana Estuary, California (ZEDLER and MADON,
personal communication, 2003). The restoration
site (8 hectares) was designed to address the im-
portance of small-scale tidal creeks on plant estab-
lishment and growth, invertebrate abundances
and fish use. The entire site receives tidal action,
and the salt marsh is divided into replicate one-
hectare treatment areas, half of which include a
network of small-scale tidal creeks and half of
which lack this heterogeneity. In a restored marsh
in Coos Bay, Oregon, creek formation was greater
at lower intertidal elevations, but vegetation de-
velopment was more rapid at high marsh eleva-
tions (CORNU and SADRO, 2002). They found that
both elevation and the gradient of the marsh sur-
face were important in determining creek forma-
tion.

As with diversity, the issue of heterogeneity is
of direct importance to restoration because most
restored salt marshes tend to lack heterogeneity.
They typically are created to be flat or very grad-
ually sloping, with much lower creek densities
than natural salt marshes. In most cases, it is an-
ticipated that tidal creeks will develop over time
once tidal hydrology has been restored to the salt
marsh. However, it is not clear how long it may
take for creeks to develop, or what may promote
the development of specific creek features. Re-
stored marshes also lack the other features that

contribute to overall physical heterogeneity, such
as hummocks, mounds, and depressions.

In planning and designing restored salt marshes
that are likely to match the functioning of natural
ecosystems, much more information about the lev-
els and scales of heterogeneity that are important
for natural salt marsh functioning is needed. Bet-
ter knowledge of the density and spatial distribu-
tion of other small-scale features across the salt
marsh landscape, including berms, natural levees,
etc., would improve our understanding of spatial
processes within natural marshes, as well as the
design of restored salt marshes. We need to con-
sider how these features vary across regions, and
how they are affected by tidal range, the relative
input of fresh water, and geomorphic conditions.
This information would help to quantify the rela-
tionships between heterogeneity and ecosystem
function. Finally, there is a need to understand
how physical heterogeneity develops in both nat-
ural and restored sites. We need to consider what
drives the creation of these features within a salt
marsh, and how long it takes for appropriate levels
of heterogeneity to develop naturally. This would
help to identify ways to promote the rapid devel-
opment of heterogeneity at restored sites.

## IV. Impacts of Exotic Species Within Restored Marshes

Exotic plants, such as Phragmites australis and
Spartina spp., are often a problem at restoration
sites because of the high level of initial distur-
bance associated with grading and other restora-
tion site preparation. These disturbances tend to
promote fast-growing exotic species that can com-
pete very effectively in resource-rich, post-distur-
bance conditions (D'ANTONIO and MEYERSON,
2002). Exotics are a concern because they can out-
compete target native species and change ecosys-
tem dynamics by affecting water cycling, nutrient
cycling, and other processes (VITOUSEK et al., 1996).
There has been little evaluation of the impacts of
exotics within the salt marsh beyond competition
with natives, but exotics can outcompete natives
and change sediment dynamics. In some cases, ex-
otic species may not be so weedy and problematic,
and in particular situations, such as severely de-
graded sites, they may be useful in ameliorating
harsh growing conditions and promoting the fu-
ture establishment of native species (D'ANTONIO
and MEYERSON, 2002). Exotic animals may also be
problematic within restored salt marshes; howev-

er, this review focuses on plants because of their primary role in creating habitat.

Compared to other ecosystems, exotic plants are not as great of a problem in salt marshes (ADAM, 2002), probably because there are few exotic species that can tolerate the stressful combination of high salinities and anaerobic conditions. In addition, natural salt marshes tend to have dense plant cover that will inhibit establishment of other plants, including exotic species. In mediterranean salt marshes, most exotic plants tend to be found in the upper part of the marsh, along the wetland-upland transition zone (MACDONALD, 1977; ADAM, 2002). This area is typically affected by ongoing disturbances, including increased local freshwater discharge, increased nutrient inputs, and increased rates of sedimentation.

Of the few exotic species that have done well in the low marsh, *Spartina* species are the most problematic, in particular *Spartina alterniflora* and *S. anglica*. *Spartina alterniflora* is native to the Atlantic and Gulf coasts of North America and has become a significant problem on the Pacific Coast of North America, in both the State of Washington and in San Francisco Bay (SPICHER and JOSSELYN, 1985; CALLAWAY and JOSSELYN, 1992; DAEHLER and STRONG, 1997; FEIST and SIMENSTAD, 2000). It was introduced accidentally into Washington around the turn of the century with the oyster industry (FEIST and SIMENSTAD, 2000) and was intentionally introduced into San Francisco Bay as an experiment in an early restoration project in the 1970s by the U.S. Army Corps of Engineers (GROSSINGER *et al.*, 1998). In Washington, concern over *S. alterniflora* focuses on the conversion of mudflats to vegetated marsh and the resulting impacts to shorebirds and the oyster industry (FEIST and SIMENSTAD, 2000). In San Francisco Bay, *S. alterniflora* is also considered a major problem due to the loss of mudflats, as well as changes in creek morphology within the marsh, and impacts on the native cordgrass, *Spartina foliosa*, due to competition and hybridization (CALLAWAY and JOSSELYN, 1992; DAEHLER and STRONG, 1997). Within San Francisco Bay, *S. alterniflora* has become a significant problem within a number of restored marshes. State and federal agencies recently purchased over 15,000 acres of salt ponds in south San Francisco Bay with plans for substantial salt marsh restoration. This will be the largest salt marsh restoration effort on the Pacific Coast, and problems associated with the potential spread of *S. alterniflora* are one of the major concerns in the resto-

ration planning effort (GOALS PROJECT, 1999). A draft environmental impact report evaluating potential control approaches for *S. alterniflora* within San Francisco Bay is currently under evaluation (see www.spartina.org).

*Spartina anglica* is another problematic invasive and is a hybrid between the native European species, *Spartina maritima*, and *S. alterniflora* which was introduced from North America. *Spartina anglica* has spread throughout Europe and has almost completely replaced the native *S. maritima* (GRAY *et al.*, 1991). In addition to the loss of biodiversity due to impacts on *S. maritima*, the loss of shorebird and wading bird foraging areas has been a major concern in England and the rest of Europe (GOSS-COSTARD and MOSER, 1988). There are additional examples of the spread of *Spartina* species in New Zealand, Tasmania, and elsewhere (LEE and PARTRIDGE, 1983; HEDGE and KRIWOKEN LORNE, 2000). While they are the target of many restoration efforts within their native ranges, these *Spartina* species represent a substantial challenge to restoration in areas where they are not native because they can easily establish into new sites if their seeds are available within the area. In this sense, they represent dispersal-limited species—those which are only limited by long-distance dispersal. Once they get to any area, they will spread rapidly and need no other changes to the ecosystem to proliferate.

On the other hand, most of the species that are found in the upper extent of the marsh, along the wetland-upland transition, are disturbance-limited species: species which need some disturbance to "natural" conditions to do well. As noted, the wetland border tends to be an area of substantial ongoing disturbance (on top of the initial disturbance associated with restoration activities), including changes to hydrological cycling, nutrient cycling, and other natural processes. In the case of hydrological cycling, many southern California wetlands receive excess freshwater inputs due to the reliance of this region on imported water. This substantially reduces soil salinities in the upper reaches of the marsh and allows for many exotic species to outcompete more salt-tolerant native salt marsh vegetation. For example, reduced soil salinities allow *Polypogon monspeliensis* to outcompete both *Salicornia subterminalis* (KUHN and ZEDLER, 1997) and *S. virginica* (CALLAWAY and ZEDLER, 1998). Another example of a disturbance-limited species is likely *Phragmites australis* (common reed) which has become a substantial prob-

lem in both natural and restored marshes along the Atlantic coast of North America (CHAMBERS *et al.*, 1999). Experimental results indicate that *P. australis* does better in salt marshes with disturbed hydrology and increased drainage (BART and HARTMAN, 2000). Restoration of natural tidal regimes can reduce the distribution *P. australis* in restored salt marshes (WARREN *et al.*, 2002).

Exotic species will remain a substantial problem for many restoration projects, and it is highly unlikely that we will ever identify any simple answer to this complex problem. The challenge lies in how to best use our effort to minimize this ongoing problem. In order to identify potential methods of control or eradication, it is necessary to understand the biology of the invading species, as well as its interactions with the restored ecosystem and native species. What works for one species will not necessarily apply for others. Similarly, opportunities for control of a single species may vary by region. For example, attempts have been made to control *S. alterniflora* in the Pacific Northwest with *Prokelisia marginata*, a planthopper that feeds on *Spartina* species (GREVSTAD *et al.*, 2003). Grazing by the planthopper reduced the biomass of *S. alterniflora* by 50% in cages in the field, although it is not clear whether planthopper populations could sustain this level of impact across larger spatial scales. This method of control cannot be used in San Francisco Bay because the planthopper also attacks the local native species, *Spartina foliosa*. Some recommendations for improving control and eradication of exotic plants at restored salt marshes include the following.

(1) Focus on early detection and rapid removal of exotics before they get well established. This is especially important for dispersal-limited species, as dispersal-limited species can be very problematic for restoration projects, and there is little that can be done to control their spread other than monitoring and direct control methods. If an exotic plant becomes widespread throughout a restoration project, it may be highly destructive to attempt eradication and these attempts are likely to fail. For example, despite significant efforts to control *S. alterniflora* in Willapa Bay, Washington, it still remains widespread within the estuary.

(2) Identify underlying disturbances that may be favoring disturbance-limited exotic species. If we eradicate a disturbance-limited exotic species from a restoration site, but the disturbances remain, the exotic is likely to re-establish over time. In this sense, we need to treat both the symptom

(exotic species) and the cause (disturbance) of the problem. Of course, this also implies that we can identify the disturbance that has caused an increase in exotic species impacts. Treating the cause of the problem may mean reducing soil nitrogen concentrations or restoring natural freshwater inputs into a local salt marsh. In other cases, it could mean diverting urban runoff away from a restoration site to avoid excess freshwater inputs that may favor exotic species. Of course some disturbances may be very difficult to remove, but we can make an attempt to reduce or localize their impacts through a range of management alternatives.

(3) Use herbicides and other destructive control methods only after carefully considering pros and cons. Attempts at control or eradication may have large negative effects due to herbicide use, impacts to other non-target species, and other impacts. Clearly there are benefits from removing an exotic species; however, in some cases eradication or control may not be possible, or the removal process may be more destructive than the benefits.

## V. Incorporation of Experimentation into Restoration

One of the largest challenges for research in restoration ecology is to identify the cause-and-effect relationships that limit the development of restored ecosystems. As noted above, trajectories may give us some insight into ecosystem development by identifying ecosystem attributes that are particularly slow to develop; however, it is only possible to identify cause-effect relationships with manipulative experiments. This identification of cause and effect is critical for improving restoration methods. Experiments could be set up within restoration projects to evaluate the effects of changes in disturbances, nutrient additions, soil manipulations, biological interactions, and other factors. With some considerations for experimental design, restoration projects offer outstanding opportunities to address these issues because in many senses, restoration projects are large-scale manipulative experiments. In the past, the challenge has been that most restoration has taken place on a trial-and-error basis, with no concern for experimental design issues, such as replication, controls, treatments, etc.

Experiments could take place within restoration on a variety of scales. At the smallest level, mesocosms and other small-scale manipulations

could be made either within a restoration project or prior to the implementation of a project. CALLAWAY and ZEDLER (1998) used this approach to evaluate the effects of freshwater inputs and tidal action on interactions between a native and exotic plant species. Mesocosm experiments could be a useful approach for testing a range of novel restoration methods, and this approach would identify methods that should be tested on a broader scale.

Larger-scale field experiments, such as those used at the Tidal Linkage to test diversity-function relationships (ZEDLER et al., 2001; CALLAWAY et al., 2003), should also be used. These would typically be incorporated directly into the design of a restoration project but use relatively small-scale treatment areas, e.g., 1–10 m. These sorts of experiments have been used widely in field ecology to test a wide range of issues, but they have not been used commonly in a restoration setting. These larger-scale experiments require greater effort but give the advantage of ensuring appropriate environmental conditions since they would be located within a restoration setting.

Finally, at the largest scale, entire restoration projects could be devoted to experimental manipulation. Outstanding opportunities exist for incorporating a range of experiments to test factors that are only possible at this large scale (e.g., the presence or absence of fine creek networks at the Model Marsh (ZEDLER, 2001; ZEDLER and CALLAWAY, 2003)). Large-scale experiments require substantial coordination and funding but allow for the rigorous testing of key factors in restoration design and implementation. These types of large-scale experiments would be more difficult to implement within the mitigation context where every acre of mitigation credit is a concern and an experimental approach may fail (unless regulators are given more flexibility in applying mitigation credits towards restoration experiments). Within restoration projects that are completed outside of mitigation constraints, the benefits from large-scale experiments are substantial. By identifying clear cause-effect relationships on the scale of an actual restoration project, the incorporation of experiments would substantially improve the design, implementation, and monitoring of future restoration efforts. It is only with these types of experiments that we will make rapid progress in improving restoration techniques.

## CONCLUSIONS

Incorporation of more science and better science into restoration planning and implementation can improve restoration efforts while increasing our understanding of ecosystems at the same time. Restoration practice will benefit because better restoration and management techniques will be developed more quickly, and science will benefit because we will gain insights into the processes that control the development of both restored and natural ecosystems.

By improving the analysis and understanding of trajectories of ecosystem development, we will improve both policy and restoration practice, and identify the time frame that is needed for the development of different ecosystem attributes. Further research evaluating the importance of species diversity for restored salt marshes is needed. Physical heterogeneity is likely to contribute to the development of ecosystem function in restored salt marshes and also needs further study. Effort toward controlling exotic plants in restored marshes should focus on identifying appropriate control methods based on the biology of the invading species and the restored ecosystem. And finally, the incorporation of more experimentation into restoration projects is needed and is the best method to identify factors that may limit or inhibit the development of restored salt marshes.

## ACKNOWLEDGMENTS

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